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ENVIRONMENTAL RISKS OF MUNICIPAL NON-HAZARDOUS WASTE LANDFILLING AND INCINERATION

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Standards Development Branch
Environmental Sciences and Standards Division
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ABSTRACT

It is the position of the Ministry of the Environment that both landfilling and incineration options be available for consideration so that a municipality or other proponent can develop the best environmental solution for its locality.

The Ministry conducted a series of technical risk assessments of two generic disposal facilities: a large-scale incinerator and modern landfill with a similar disposal capacity of 6.6 million tonnes over 20 years.

Potential emissions from these hypothetical facilities were assessed for their potential impacts on human health and on the environment. Negligible effects are predicted for facilities that meet stringent requirements and standards for design, operation and pollution control. Therefore, concerns for human health and the environment should not limit the selection and approval of incinerators or landfills. Specifically:

- No significant human health effects (those being cancer, lung disease, nerve damage or reproductive effects) are likely in a typical suburban community located near an incinerator or a landfill. Under certain conditions, nuisance problems linked to malodorous compounds may affect air quality close to a landfill.

- An ecological risk assessment predicts that water and sediment quality near an incinerator or landfill will meet Ministry guidelines for the protection of aquatic life.

- Direct or indirect impacts to the terrestrial environment, vegetation or wildlife resulting from incinerator or landfill emissions are not anticipated to be significant. The main differences in terrestrial impacts between the two waste disposal methods relate to the amount of land used and to the production of nitrogen oxides.

EXECUTIVE SUMMARY

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EXECUTIVE SUMMARY

In the hierarchy of preferred waste management options, incineration and landfilling follow the 3Rs -- waste reduction, reuse and recycling. Despite the success of waste diversion efforts, however, residual wastes still require disposal.

Up-to-date information is needed on the potential environmental and human health impacts of the incineration and landfilling for municipal waste disposal. To obtain this information, the ministry conducted a series of technical risk assessments of two generic disposal facilities: a large-scale incinerator and a modern landfill, each with a similar disposal capacity of 6.6 million tonnes over 20 years.

Section 2 of this report provides summaries of the main assumptions, methods and conclusions of the full technical studies for the landfill and incinerator risk assessments. The full technical reports on the health impacts of the generic incinerator and landfill are attached as Technical Report Parts A and B respectively. Technical Report Parts C and D, respectively, provide the full technical reports on the aquatic and terrestrial effects of both technologies. Glossaries of technical terms used in the risk assessments are included as Technical Report Part E.

The risk assessments show that properly designed and operated municipal waste incinerators and landfills should not have a significant effect on human health and the environment. Beyond these analyses, new municipal waste incinerators and landfills will be required to meet Ontario's stricter *Guideline A-7* and *Landfill Standards* guideline requirements. Consequently, new facilities will present even less risks than those identified in these analyses.

1.1 Effects on human health

The human health assessments show that no adverse health effects (including cancer, lung disease, nerve damage and reproductive effects) are expected to result from long-term exposure to emissions from either type of facility.

Estimates of the total lifetime incremental cancer risk for a maximally exposed individual, from long-term exposure to incinerator emissions, range from 4.7×10^{-8} to 2.3×10^{-7}. Estimated exposure to emissions of non-carcinogenic chemicals were below generally accepted levels of concern.

Estimates of the total combined cancer risk from long-term exposure to landfill emissions containing volatile chlorinated chemicals range from 4.0×10^{-6} to 1.0×10^{-5}.

Landfills could result in complaints about odours and subjective symptoms but no adverse health effects are expected.

1.2 Effects on the aquatic environment

Water and sediment quality around an appropriately designed and operated landfill or incinerator is expected to meet ministry guidelines for aquatic life. However, mercury and dioxin/furan levels after 20 years of incineration operation, might slightly exceed U.S. EPA water quality guidelines that were developed to protect wildlife which consume fish.

1.3 Effects on the terrestrial environment

Estimated chemical emissions from either a landfill or an incinerator will not cause adverse effects on nearby vegetation, wildlife or other terrestrial biota. Further assessment may be required to determine the risk to wildlife that consume soil-dwelling organisms at sites where the background levels of persistent toxic chemicals in the soil are higher than normal.

While the operation of a landfill requires at least four times the land space of an incinerator, upon closure the area can be reclaimed as green space, partially compensating for loss of habitat.

Incinerators emit more nitrogen oxides than do landfills. However, total emissions of nitrogen oxides from incinerators would be a small compared to the total nitrogen oxide emissions in Ontario. Given that small contribution, it was not possible to statistically separately evaluate the potential adverse effects on sensitive upland hardwood forests which may already be under stress from current nitrogen oxide levels.

A detailed comparative analysis of greenhouse gas emissions from landfills and incinerators was not within the scope of this study. However, when estimated emissions of CO_2 and methane are converted to greenhouse gas equivalents, and the uncertainty associated with emission estimates are taken into consideration, both disposal methods are roughly equal with respect to the release of greenhouse gases.

RISK ASSESSMENTS

2.1 Introduction

Ministry staff have undertaken extensive technical studies to quantify and evaluate the potential adverse effects of incineration and landfilling of municipal solid waste, on human health and the natural environment. The study provides interested parties with baseline technical information needed to support the decisions on waste disposal.

Risk assessments were conducted on human health effects, terrestrial effects and aquatic effects. The release of toxic chemicals to the environment was the prime concern, but other risks were evaluated including: odour and nuisance problems, the loss of habitat and greenhouse gas emissions.

The assessment of potential environmental and health effects followed accepted scientific procedures used by risk assessment experts and government agencies in Canada and the United States. Screening protocols and sophisticated modelling simulated a range of environmental exposures and corresponding risk levels.

Realistic scenarios concerning disposal facilities, locations, emissions, populations affected and a wide range of environmental and human health effects were examined. For each scenario, staff estimated potential chemical exposures and weighed them against stringent toxicological criteria designed to protect human health and the environment.

2.2 Sources of information

The ministry maintains extensive scientific, technical and regulatory information on the environmental and human health effects associated with the disposal of municipal solid wastes. The assessments provided in Parts A through D of this technical report include detailed bibliographies and appendices.

The assessments were based on currently available information in the Ministry's data bases. Wherever possible, Ontario data were used. Where additional information was required, it was obtained from the scientific literature or other sources available to the Ministry.

It was decided that new field or raw data would not be developed for the study.

2.3 Risk assessment assumptions

Risk assessment is a formal, scientific process to analyse the potential for harm following exposure to chemicals or other agents. A number of basic assumptions are used to define and set the limits of the study, and the findings are valid only in accordance with these underlying assumptions.

For the purpose of these assessments, a hypothetical incinerator and a landfill of equivalent size (capable of disposing of 6.6 million tonnes of municipal waste over 20 years), were assumed to be sited in Ontario. The assessments use the same data bases for meteorological, population, topographical, land area and surface water information. Because extensive, site-specific data had been collected recently in a series of technical reports prepared in support of a southern Ontario landfill proposal, these reports were the major source of data for the studies. A population base of 550,000 residents was used for the study.

The incinerator and landfill assessments were designed to be comparable, but specific assumptions were needed to account for differences between the waste disposal methods. For example, the potential effects of incineration commence immediately with startup and end when the facility shuts down. In contrast, landfill emissions peak later in the operating life of the facility (after about twenty years) and decline slowly following closure.

While risk assessment is a rigorous scientific discipline, it has certain limitations. All risk assessment procedures contain uncertainties due to, for example, limited information on the release of specific contaminants to the environment, their exposure pathways and toxicological effects. Therefore, the exercise yields only an estimate of the risk and not the actual effect associated with a specific proposed or existing facility.

To compensate for uncertainties, "worst case" or "reasonable worst-case" chemical exposures area assumed. The use of higher-than-expected exposure conditions emphasizes any potential problems and identifies potentially dangerous chemicals or sensitive exposure pathways for more detailed study.

In most cases, this will result in overestimating the actual risks. A projected exposure level that exceeds an environmental or human health protection standard represents a warning flag, rather than an actual risk or effect.

For the purposes of this report, background levels of chemicals were not evaluated because they are site-specific and variable, depending on the presence of natural and manufactured sources. When assessing a particular proposed site the local background levels will contribute to the overall environmental health risk.

The primary sources of operating data and major assumptions are described for the landfill and incinerator, below. All other assumptions used in the analyses (such as deposition rates or health criteria) are noted elsewhere in the attached assessment reports.

The incinerator

Wastes have been burned in Ontario for a long time. The technology has evolved from burning piles of refuse in open pits to highly-engineered plants that incorporate extensive computer controls, pollution control equipment and continuous emission monitoring. There are two main types of incinerators widely used around the world: mass burn units and modular starved air units.

- The hypothetical incinerator used for the analyses is a state-of-the-art facility with well operated, advanced pollution control equipment.

- The incinerator would have disposal capacity of 6.6 million tonnes of municipal solid waste, over an operational life of 20 years (to correspond with the capacity of the landfill).

- The estimates of air emissions were based on stack monitoring data for a medium-sized and a larger facility, burning waste at a 400 and 1500 tonnes per day, respectively.

- The analysis used the latest emission testing data from incinerators operated in Brampton, Ontario, and Hartford, Connecticut. Both facilities burn municipal wastes: the Hartford facility employs mass burn technology, while the Brampton facility uses starved air. Both plants have modern pollution control equipment, including wet scrubbing, lime injection and bag filtration systems. Extensive data derived from emission monitoring at both facilities was compiled to determine the most significant pollutants for further consideration.

- The dispersion of incinerator air emissions was calculated using the ministry's standard modelling approach.

- Neither of the incinerators on which much of the analysis was based currently employs nitrogen oxide controls or adds activated carbon to the scrubber (used, primarily, to reduce mercury and dioxin/furan levels). As a result, the emissions and associated risks are higher than any new incinerator facility in Ontario which would have to meet the province's more stringent approval and operating standards.

The landfill

Disposal of municipal refuse into landfills has also been carried out in Ontario for a long time. As a technology, it has evolved from simply dumping waste onto the ground to engineered facility designed and operated to protect the environment.

The landfill studied in the risk assessment is generic, but the analysis is based on data generated during the development of landfill proposals for a Southern Ontario area. This provided an up-to-date and extensive data base which had been reviewed recently by Ministry staff involved in the landfill approval process.

- The hypothetical landfill used for the analyses is a state-of-the-art facility with advanced pollution control equipment that is well operated without failure.

- The landfill would have a disposal capacity of 6.6 million tonnes of municipal wastes, over an operational life of 20 years (to correspond with the capacity of the incinerator).

- The impact assessment for the landfill assumes an environmental emission period extending 20 years beyond closure.

- Data on terrestrial and aquatic effects are based on a landfill proposed in reports to the Interim Waste Authority.

- Leachate is collected, treated on-site and then discharged to a municipal sewer system. The stormwater is collected and treated in a settling pond prior to discharge to a local water course.

- Landfill gas collection starts in the fourth year of operation and continues 20 years after closure. The gas collection system recovers 70 per cent of the gas produced, while the remaining 30 per cent escapes to the atmosphere. The collected gas is burned in a flare, without any pollution control system.

- Estimates of landfill gas emissions are based on data derived from the Britannia Road landfill in the Region of Peel, Ontario.

- The dispersion of landfill air emissions was calculated using the Ministry's standard modelling approach.

2.4 Environmental fate of emissions

The levels of chemical contaminants to which humans and wildlife are exposed depend on the environmental fate of incinerator and landfill emissions. To estimate exposures, mathematical models were used to simulate the dispersion of emissions into the environment surrounding each waste facility. The models estimated chemical concentrations in air, soil, water, sediment and the food chain. Southern Ontario weather data were used in the modelling.

The environmental pathways of incinerator and landfill emissions to humans and wildlife are illustrated in Figures 1 to 4 below.

Figure 1: Aquatic exposure pathways

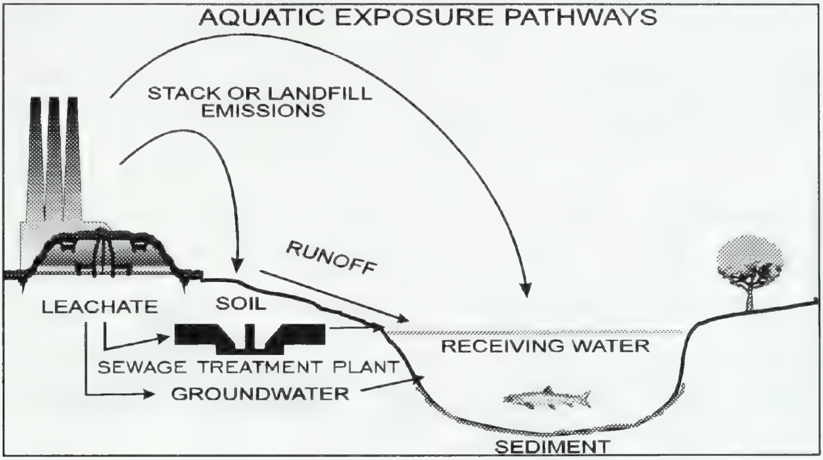


Figure 2: Terrestrial exposure pathways

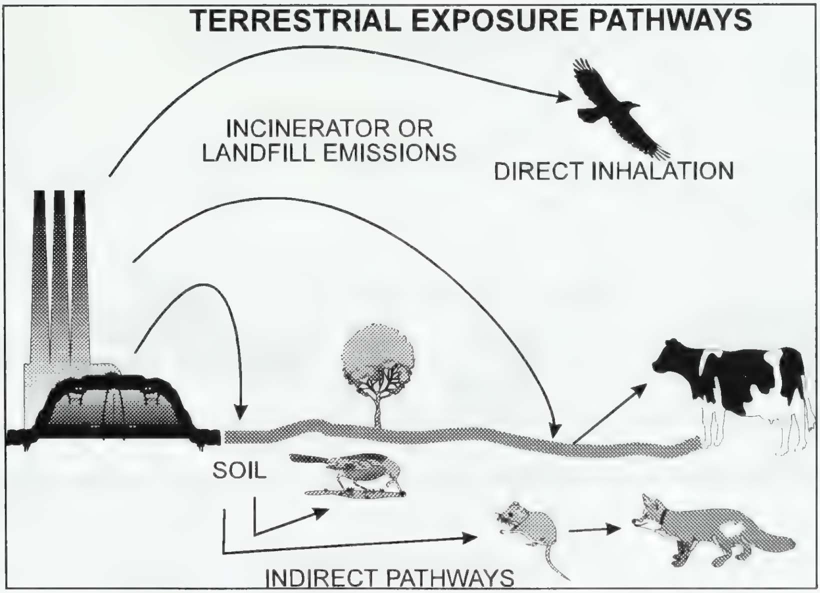

TERRESTRIAL EXPOSURE PATHWAYS
INCINERATOR OR LANDFILL EMISSIONS
DIRECT INHALATION
SOIL
INDIRECT PATHWAYS

Figure 3: Incinerator exposure pathways

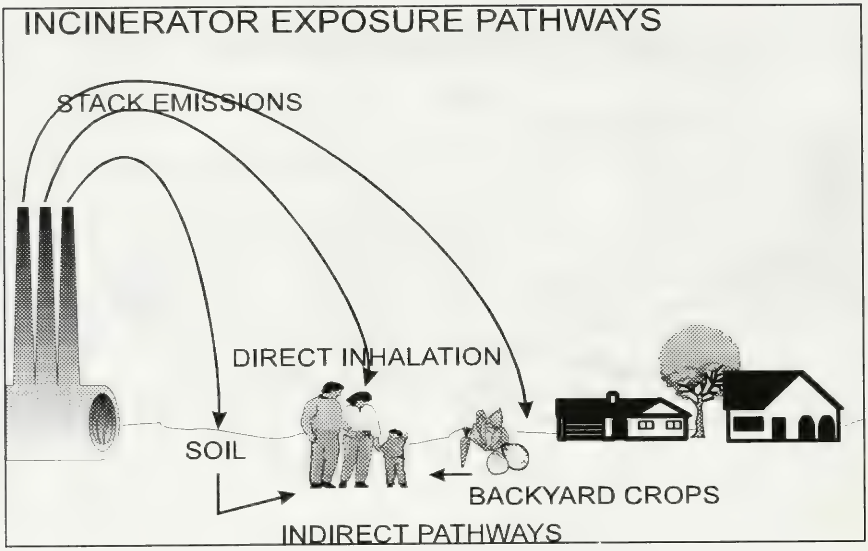
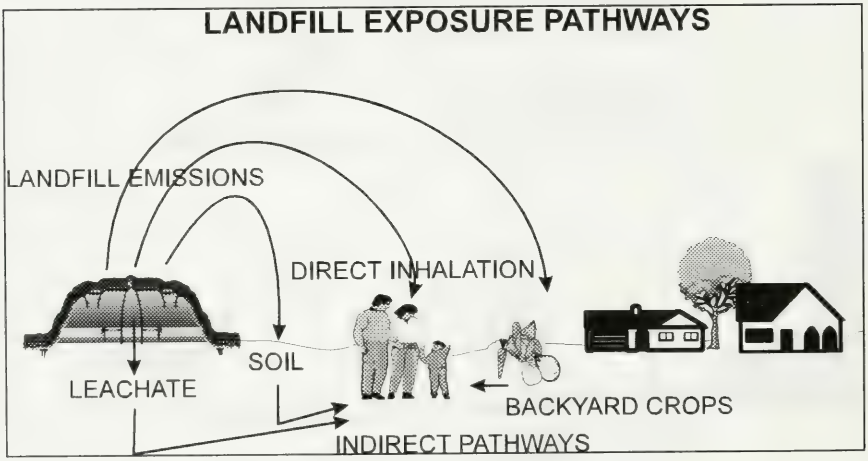


Figure 4: Landfill exposure pathways



The incinerator

A municipal solid waste (MSW) incinerator emits gases and small particles, including trace levels of metals and organic chemicals into the atmosphere from a smoke stack, usually more than 30 metres above ground level. These emissions are transported downwind and dispersed, according to weather conditions and the local terrain. The bulk of the emissions are spread widely over long distances, contributing to overall provincial air pollution. However, some of the emissions reach ground level within a few kilometres of the stack, creating an area of maximum ground-level air concentrations. This area of maximum exposure may include urban communities, rural areas, forests, lakes and streams.

Incineration produces both fly ash, the small particles filtered out by the air pollution control equipment, and bottom ash as solid waste residues. Bottom ash is considered nonhazardous and can be landfilled along with municipal solid waste. The chemicals in the ash are not expected to migrate or "leach" from the landfill by water drainage because of their low mobility in soils and their containment within the clay liners of a well-designed landfill.

Fly ash is tested for leachate toxicity. If tests show it is toxic, it must be disposed of in a secure, hazardous waste landfill.

The landfill

Over the course of its operating life and following closure, a municipal solid waste landfill releases gases and particles to the air through out-gassing, and to the soil and groundwater, through leaching. Traces of organic compounds and metals are found in the emissions.

Most of the organic gases, such as methane in the air emissions, are captured and burned-off in flares; with the remainder dispersed downwind. Compared to an incinerator, the area of maximum exposure would be much closer to the source, near the property line of the landfill.

Some landfill chemicals are mobile in leachate and must be collected and treated. Leachate that escapes a faulty landfill may contaminate groundwater and surface water.

2.5 The assessment of effects on human health

Potential human health effects associated with the operation of the hypothetical incinerator and landfill were assessed by:

(1) screening lists of possible chemical emissions and preparing a shorter list of priority chemicals;

(2) estimating levels of environmental contamination by these priority chemicals to which people might be exposed;

(3) comparing the exposure (intake) to established health protection criteria.

To estimate human exposure, information was needed on the chemical concentrations in air, water, soil and food. There is no single correct level for each concentration, but a range of possible values that will vary according to location and environmental condition.

The conventional method for assessing health impacts is to use a "worst-case" level for each chemical of concern and make a single estimate of maximum risk. For this study, more realistic values were used for some of the exposure concentrations to make a "reasonable worst-case" exposure scenario for each priority chemical.

A second, more sophisticated method was used to estimate human exposure and health risks. A computer model, using a "Monte Carlo" methodology, analysed the range of possible levels for each exposure variable and generated a corresponding range of risk estimates for each chemical. This provided more information (for example, midpoint and 95th percentile risks) than the conventional method that examines only one scenario for each chemical.

The reasonable worst-case exposure scenarios were also included in the Monte Carlo analysis. This ensured that the findings of both methods could be compared to published risk assessments.

The most up-to-date federal, Ontario and U.S. human health protection information were used to assess health risks. Several different health effects were considered, including cancer. For chemicals that cause cancer (carcinogens), human exposures were converted into a lifetime cancer risk. Many jurisdictions consider a lifetime risk of one in one hundred thousand (0.00001 or 10^{-5}) or less to be negligible or not significant.

For chemicals that can cause other toxic effects – such as lung disease, birth defects or nerve damage – chemical exposures were compared to existing health protection standards to calculate an exposure ratio. A ratio of less than one (1.0) means that no adverse health effects are expected.

Cancer risks or exposure ratios were calculated for each chemical. To assess the combined overall health impact, the cancer risks and exposure ratios were added together for different chemicals with similar toxic effects.

In this report, the term "dioxin" refers to the large but closely related dioxin and furan family of compounds. Although certain dioxin compounds may cause cancer, it is believed that a threshold or minimum level is necessary for this occur. Studies indicate that the impact of dioxin on the reproductive system may be just as significant as its carcinogenic effects. Therefore, dioxin toxicity was evaluated using exposure ratios and a stringent health protection standard.

The lifetime cancer risk for the community (of approximately 550,000 residents) living near the facility was also estimated using the Monte Carlo method. The midpoint and upper limits (that is the 95th and 97th percentile) of population risk were expressed as the number of cancer cases that might be expected due to chemical emissions during the operating life of the facility.

The possibility of nuisance odours was evaluated for landfilling but not incineration.

The incinerator

Since incineration generates a wide variety of chemicals, many of them in trace amounts in the off-gases and ashes produced, it would be a practical impossibility to evaluate them all. This assessment focussed on the chemicals in stack emissions that are either abundant or can cause serious human health effects. A master list of 130 incinerator chemicals was reduced to 70 chemicals for a second screening step.

The second screening identified those chemicals that may be present in significant levels or represent health risks to a maximally exposed individual breathing air contaminated by incinerator emissions. This screening step resulted in the compilation of a short list of 15 priority chemicals for detailed study (see Table 1 in the Appendix). Together, these priority chemicals accounted for 99 per cent of the risk related to the inhalation of toxic emissions. Cancer, lung disease and nerve damage are the main health hazards posed by these 15 chemicals.

The dispersion of the priority incinerator emissions into air and the surrounding community was simulated using mathematical models. The airborne concentrations were calculated for the area of maximum exposure near the incinerator. In turn, these levels were used to estimate the resulting contamination of local soil and of fruits and vegetables grown in backyard gardens.

People would be exposed to the chemicals primarily by breathing air (inhalation) and eating garden produce (ingestion). Chemical exposure may also result from skin contact with contaminated soil and from the accidental ingestion of soil. Because a suburban lifestyle was assumed, other exposure routes not directly affected by incineration emissions, were considered insignificant.

The typical suburban diet would consist primarily of supermarket foods, treated tap water and other sources unaffected by the incinerator emissions. Local sources of food and water may be contaminated, but would represent a smaller part of the average suburban diet (as compared to the diet of someone who lived on a farm or who consumed significant amounts of game or sport fish).

The results of the human health assessment are summarized in Technical Report Part A.

For the "reasonable worst-case" exposure scenario, the main findings are as follows:

- The combined cancer risk was 8.3×10^{-8} (*i.e.*, 1 in 12 million). Almost all of the lifetime cancer risk was related to inhalation; other exposure routes added little to the risk.
- The combined exposure ratio (the ratio of the estimated exposures to published health protection limits) was 0.1 for those priority chemicals with effects other than cancer. A ratio more than 1.0 is considered to be above accepted exposure limits.
- The intake of dioxin was less than one per cent of the recommended exposure limit. Dioxin intake was due mostly to indirect exposure from backyard garden produce and soil.
- For lead and mercury, soil and garden produce contributed over half of the exposure ratio.

The main findings of the computerized *Monte Carlo* exposure assessment are as follows:

- The combined cancer risk ranged from a midpoint value of 4.7×10^{-8} (one in 21 million) to an upper limit of 2.3×10^{-7} (one in 4.4 million).
- The combined exposure ratio ranged from a midpoint value of 0.06 to an upper limit of 0.33.
- The exposure ratio for dioxin ranged from a midpoint value of 0.06 to an upper limit of 0.15.
- Estimates of population health risk indicate no cases of cancer would be expected due to exposure to incinerator emissions.

These results suggest lifetime cancer risks are negligible for individuals and populations living near the incinerator. For effects other than cancer, the incinerator emissions made no significant contribution to everyday exposures. Overall, chemical emissions from the incinerator would not impact adversely, individuals or populations living around the facility. These findings are consistent with other scientific studies which have not predicted adverse health effects from the operation of incinerators.

The health risks of landfilled incinerator ash and the groundwater migration of chemicals in the ash could not be assessed in as great detail as the air emissions. Other studies have indicated that the more hazardous chemicals in bottom ash are unlikely to migrate from a modern landfill, even if the liner fails. Bottom ash and its leachates have not been evaluated for all chemicals, so there may be some chemicals with potential effects, although the possibility appears remote. This could be a topic for consideration in future research.

The landfill

The health impact assessment of air emissions from a landfill followed the same protocols developed for the incinerator assessment. In addition, landfill air emissions were evaluated for the possibility of nuisance odours and cancer risks were estimated for landfill leachate reaching a groundwater drinking supply.

To select priority chemicals for more detailed study, screening was conducted for 68 substances that could be present in air emissions and for 170 in the leachate. Table 2 lists the 18 priority chemicals that were selected.

All the volatile organic chemicals present in the leachate are carcinogens. Other substances, such as iron, zinc and lead, are associated with other significant health effects. Three chemicals are associated with nuisance odours: ethyl mercaptan, methyl mercaptan and hydrogen sulphide.

Exposures were estimated for the chemicals in air, groundwater, soil and backyard garden produce. Vapour movement through soil and basement infiltration were not considered.

The main findings of the health effect assessment for landfill emissions are as follows:

- The combined cancer risk ranges from 4×10^{-6} (1 in 250,000) to 1×10^{-5} (1 in 100,000). None of the individual chemical risks exceeded 1×10^{-5} (1 in 100,000).

- The combined exposure ratios (that is, the ratio of the estimated exposures to published health protection limits) were generally around or below 1.0 for other toxic effects.

- Risks associated with drinking water that may be contaminated by landfill chemicals (e.g., vinyl chloride) were insignificant; i.e. less than 1×10^{10} (1 in 10 billion). For the reasonable worst case exposures, risks were below 1×10^{-6} (1 in a million).

- Estimates of population health risk suggest no cases of adverse health effects resulting from exposure to the mixture of chlorinated organic chemicals.

- Some exposure ratios above one (1.0) were found for particle-bound iron and odour compounds.

Most of the projected health risks and exposures were well within acceptable levels for most scenarios. Some exposure ratios for particle-bound iron were four to five times over the health protection limit. These are not necessarily significant because the health protection limit includes a margin of safety and the ratios were for extreme exposures that would rarely occur under actual operating conditions. The conclusion is that chemical emissions from landfills are not expected to harm individuals or populations living around the facility.

Complaints about nuisance odours, arising from the release of ethyl mercaptan and methyl mercaptan, could be expected. Sensitive individuals may detect other odorous chemicals. It is not possible to predict fully the extent of odour problems, because these depend on weather patterns dispersing the chemicals through the local community. In addition, certain chemicals may have a stronger odour when present in a mix of other odorous chemicals. Each year, ministry officials receive a number of complaints about landfill odours from people living near these facilities. Studies of landfills outside Ontario also report odour complaints and other subjective symptoms, but do not confirm any adverse health effects.

Conclusions

The health effect assessments predict that incinerators and landfills that are designed and operated as state-of-the-art facilities have negligible effects on human health. Specifically, no adverse health effects (cancer, lung disease, nerve damage or reproductive effects) are likely to occur in a typical community near a modern landfill or incinerator. In certain circumstances, nuisance problems from malodorous compounds may affect some people near a landfill.

2.6 The assessment of effects on the aquatic environment

Both surface water and bottom sediments were evaluated for potential contamination. In the initial assessment, worst-case exposure was assumed for each chemical of concern. Consequently, a calculated exposure that exceeded an aquatic protection limit would identify a potential risk and indicate the need for additional assessment.

Potential aquatic effects from the hypothetical incinerator and landfill were assessed as follows:

- 83 incinerator chemicals and 90 landfill chemicals were identified for assessment;
- exposure concentrations were estimated for the chemicals in surface water and sediments;
- the exposure concentrations were compared to existing aquatic protection limits.

The aquatic protection limits used were: the Provincial Water Quality Guidelines or Objectives, the Provincial Sediment Quality Guidelines, the Canadian Water Quality Guidelines and/or New York State and U.S. EPA wildlife criteria developed for the Great Lakes. These are stringent limits, generally derived using a safety factor of 10 or more. The lowest concentration of a substance capable of producing an adverse effect is determined and the level is then divided by 10 to set the protection limit.

Water quality guidelines were unavailable for some chemicals, so surrogate limits were derived using larger safety factors. An exposure ratio was calculated for each chemical by dividing the predicted concentration in water or sediments by the appropriate guideline. An exposure ratio less than one (1.0) means no aquatic environmental effects are expected.

The incinerator

Chemicals in airborne emissions from an incinerator could reach local surface waters by direct deposition and indirectly by deposition onto land and then runoff into the water body. A mathematical model was used to simulate the air dispersion, deposition and runoff of the chemical emissions.

The surface area of the study site was comprised of 96 per cent land and 4 per cent water. It was assumed that for each chemical, 40 per cent of the deposition onto land would reach the water. Thus, land runoff accounted for 90 per cent of the total chemical input to surface water, with 10 per cent from direct deposition.

The chemical concentration in sediments was assumed to equal the concentration in soils after twenty years of accumulation with no chemical dilution, breakdown or losses. This is a worst case scenario.

The main findings of the assessment of aquatic effect for incinerator emissions are as follows:

For the 83 incinerator chemicals, there were no estimated exceedences of water quality guidelines developed for the protection of aquatic life. For mercury and dioxins/furans there was potential for exceeding the guidelines to protect consumption of sport fish by anglers (mercury) and consumption of fish by wildlife (mercury, dioxins/furans) which depend upon fish for food. A second assessment using more realistic assumptions still indicated some potential to exceed wildlife and sport fish guidelines.

Sediment guidelines were available for 19 chemicals. Chemicals without guidelines were not assessed further. After 20 years of accumulation, sediment concentrations were expected to comply with current guidelines with the exception of manganese. This is apparently due to the higher background concentrations in soil compared with sediment. The projected manganese concentrations are not expected to impair sediment quality.

The landfill

The assessment of the effect of landfill emissions on the aquatic environment considered those chemicals found in air emissions, mobile in soils or often-detected in leachates. Three pathways for inputs to surface water were examined: direct deposition of air emissions, leachate discharges and groundwater discharges.

Deposition to surface water was estimated for dispersed air concentrations of landfill gases. The gases are volatile, so land runoff and sediment accumulation were considered insignificant. Also, a modern landfill would have a stormwater runoff retention pond.

Studies have shown surface water contamination caused by poorly-treated leachate draining from some Ontario landfills. Since the present assessment assumed a modern landfill equipped with pollution controls, the leachate concentrations were lowered by a factor of one thousand. This is a conservative reflection of the contaminant removal provided by on-site treatment and at the municipal sewage plant to which the treated leachate would be pumped.

Reasonable worst case estimates were selected to represent the groundwater concentrations. Chemical breakdown and other losses such as evaporation were assumed to be negligible, except as noted above for leachate. Leachate and groundwater concentrations were reduced by a factor of ten times to calculate surface water concentrations (to account for dilution in the receiving water body).

The main findings of the aquatic impact assessment for landfill emissions are as follows:

- Groundwater discharges resulted in surface water concentrations that did not exceed guidelines.
- After accounting for leachate treatment and dilution, leachate chemicals in surface water were at least one hundred times lower than the guidelines.
- The deposition of air emissions to surface water resulted in water concentrations above the guidelines for four chemicals: 3-methylheptane, nonane, octane and toluene.

The surface water concentrations predicted for 3-methylheptane, nonane, octane and toluene are for a worst-case exposure scenario. These volatile chemicals are not considered persistent and would rapidly evaporate into the air. Realistically, water concentrations are expected to be much lower than the projected levels. In addition, only the guideline for toluene is an established limit. The guidelines for the other three chemicals are surrogates developed using large safety factors. The surrogate guidelines may overestimate aquatic risk since other closely related chemicals have less stringent established guidelines. A second screening approach for these chemicals predicted what the concentrations should be, based on the chemical properties in water. The surface water concentrations estimated in the screening assessment are at least ten times lower than the predictable toxic concentrations. Considering all the evidence, the conclusion is that air emissions of landfill chemicals are unlikely to impair surface water quality.

Conclusions

Most chemical emissions from incinerators and landfills are predicted to have no adverse effects on surface water quality.

Mercury and dioxin emission from incinerators were selected for more detailed assessment due to their potential accumulation in sediment. There is some potential for exceeding the guidelines for the protection of consumption of sport fish by anglers (mercury) and consumption of fish by wildlife which depend upon fish for food (mercury, dioxins/furans), after twenty years of facility operation.

2.7 The assessment of effects on the terrestrial environment

Potential effects on the terrestrial environment from the hypothetical incinerator and landfill were assessed as follows:

- animal and plant species, common to southern Ontario, were selected for study;
- a detailed food pathway model was constructed to estimate chemical exposures for the study species;
- a master list of chemical emissions was screened to develop a short list of priority chemicals;
- the exposures by the study species for the priority chemicals were estimated using the food pathway model;
- the exposures were compared to known toxic or safe exposure levels, or were converted to cancer risk estimates; and,
- potential physical effects were assessed through literature reviews and narrative descriptions.

For practical reasons, several assumptions were necessary to limit the scope of the study. All animal and plant species could not be considered, so representative, “receptor” species were selected for detailed study. Only species common to southern Ontario were selected (most of the province's waste disposal facilities are sited in southern Ontario). While potential effects could be spread over hundreds of years, the study included only the time period when peak effects are likely.

Seven key receptors were selected: plants (grasses and leafy plants), cattle, voles, foxes, hawks, earthworms, and woodcock (a bird species). A detailed food pathway model was constructed: for plants exposed to chemicals by direct contact with air emissions and by uptake from contaminated soil; and for animals exposed by breathing air, drinking water and eating soil, vegetation or other smaller animals.

Earthworms were assumed to be eaten by woodcock in a separate assessment of chemicals persistent in the soil.

Next, a master list of chemical emissions was screened to produce a list of priority chemicals (see Table 3). The priority chemicals were selected by comparing the amounts in emissions and in the environment to: analytical detection limits, air quality guidelines (including human health limits and cancer risks), Ontario loadings or background soil concentrations.

The food pathway model was then used to calculate detailed estimates for the study species and priority chemicals. For most of the chemicals, the exposure estimates were compared to toxic or safe levels in order to calculate an exposure ratio. The toxic or safe levels were determined from a review of the scientific literature. No terrestrial environmental effects are expected for a chemical with an exposure ratio of less than one (1.0).

For a carcinogen, the receptor's exposure (intake) was converted to a lifetime cancer risk using human health risk factors. To assess the overall cancer risk, the risk levels for all the carcinogens were added together. A combined total risk of one in one hundred (1:100) was selected as meaningful for wildlife populations (consistent with an approach used in New York state). While this is higher than acceptable lifetime risk for human health, it was intended for wildlife populations, not for individual people.

Other terrestrial effects were evaluated including: greenhouse gas emissions, habitat loss, acid gases, erosion, drainage and water table effects, dust, noise, litter and pest species. These are discussed in the Technical Report Part D.

Finally, a review was conducted of ministry soil and vegetation surveys and complaint investigations around landfills and incinerators.

Generally, birds received the highest exposures and were the most sensitive study receptors. Mercury was the only substance for which some relevant toxicity data were available for all the study receptors. For the other chemicals, assumptions were made to apply toxicity data to the study receptors. Little is known about the toxicity of such chemicals as 2-methylfluorene and bromodichloromethane, so conclusions about these chemicals are tentative.

The incinerator

The main findings of the assessment of terrestrial effects from incinerator emissions are as follows:

- The exposure ratios and cancer risks were less than the levels of concern, except for mercury from the incinerator.

- The estimated levels of incinerator mercury in the hawk and woodcock suggested there might be possible effect levels for some sensitive bird species (with an exposure ratio 1.56). The estimated amounts of incinerator mercury in hawks was for a worst-case exposure. Actual exposure levels are likely to be lower than the projected levels. Also the terrestrial protection limit assumes that all the mercury is present in the most toxic form. In reality, this is usually only a small portion of the total mercury exposure. Thus significant measurable effects on bird populations are not expected. A more detailed, realistic assessment would be required to better estimate mercury risks to sensitive birds (for example, raptors that prey on small animals, as well as birds that consume soil-dwelling animals such as worms).

- Modern incinerators and landfills with gas collection systems make roughly equal contributions to the greenhouse effect. Incinerators produce carbon dioxide as a major byproduct of combustion. In landfills, lower carbon dioxide production is augmented by methane gas production.
- Incinerators produce more nitrogen oxides, an acid gas, than landfills. A large-scale switch from landfills to incinerators would increase the gas emissions (although this would be a relatively minor source when compared to other sources in Ontario). The concern is that some upland hardwood forests may already be under stress from excess nitrogen oxide loadings and increased amounts may hasten the suspected damage.

The landfill

The main findings of the terrestrial impact assessment for landfill emissions are as follows:

- The exposure ratios and cancer risks were less than the levels of concern.
- For landfill chemicals, the combined total risk of cancer to the hawk was 4:1000. The total combined cancer risk for the landfill chemicals was below the level of concern, set at 1:100.
- Landfills with gas collection systems and modern incinerators make roughly equal contributions to the greenhouse effect. Landfills convert waste to carbon dioxide and also produce methane. Both gases contribute to the greenhouse effect, but methane is a more potent greenhouse gas (21 to 24.5 times more potent than carbon dioxide).
- Landfills occupy and alter more land than do incinerators. Most of this land can be returned to greenspace when the facility closes. Although the ecosystem at a landfill site is less diverse than its original state, the site can provide wildlife habitat and urban parks.

Conclusions

In a review of ministry surveys and complaint investigations, compiled over 20 years, no evidence was found of adverse effects from landfills or incinerators that were well built and well operated. However, adverse effects were found at facilities exhibiting design faults (for example, the lack of gas collection and venting systems), failures in treatment systems, or poor drainage.

Assuming modern, well-run facilities (including collection systems for landfill gases and leachates and advanced pollution controls on incinerators), the chemicals emitted by these facilities are anticipated to pose an insignificant risk of direct adverse effects on terrestrial life forms.

2.8 Risk assessment conclusions

Potential emissions from a hypothetical incinerator and landfill were assessed for their potential effects on human health and the natural environment. Negligible effects are predicted for facilities that meet stringent requirements and standards for design, operation and pollution control. Specifically:

- No significant human health effects (those being cancer, lung disease, nerve damage or reproductive effects) are likely in a typical suburban community located near an incinerator or a landfill. Under certain conditions, nuisance problems linked to malodorous compounds may affect some people close to a landfill.

- An ecological risk assessment predicts that water and sediment quality near an incinerator or landfill will meet Ministry guidelines for the protection of aquatic life.

- Direct or indirect impacts to the terrestrial environment, vegetation or wildlife resulting from incinerator or landfill emissions are not anticipated to be significant. The main differences in terrestrial effects between the two waste disposal methods relate to the amount of land used and to the production of nitrogen oxides.

TABLES

Table 1: Incinerator chemicals assessed in detail for human health risks

Carcinogens:	Other toxic chemicals:
• chromium • benzene • cadmium • vinyl chloride • arsenic • dioxin	• nitrogen oxides • silicon • hydrogen chloride • mercury • sulphur dioxide • particulate emissions • iron • tin • lead

Table 2: Landfill chemicals selected for detailed health risk assessment

Air emissions Carcinogens: • benzene • vinyl chloride • bromodichloromethane • 1,1-dichloroethane • 1,2-dichloroethane • 1,1-dichloroethylene • 1,1,2,2-tetrachloroethane • 1,1,2-trichloroethane • trichloroethylene • methylene chloride • chloroethane	Air emissions (continued) Malodorous chemicals: • hydrogen sulphide • ethyl mercaptan • methyl mercaptan Particle-bound metals: • iron • zinc • lead Leachate emissions Carcinogens: • vinyl chloride

Table 3: Priority chemicals selected for detailed terrestrial risk assessment

Incinerator emissions:
- benzene
- dioxin
- mercury
- 2-methylfluorene
- nitrogen oxides
- hydrochloric acid

Landfill emissions
- benzene
- bromodichloromethane
- 1,2-dibromoethane
- 1,1-dichloroethane
- 1,2-dichloroethane
- 1,1-dichloroethylene
- hydrogen sulphide
- ethyl mercaptan
- methyl mercaptan
- 1,1,2,2-tetrachloroethane
- 1,1,2-trichloroethane
- trichloroethylene
- vinyl chloride

TECHNICAL REPORT – PART "A"

ASSESSMENT OF THE POTENTIAL HEALTH RISKS ASSOCIATED WITH CHEMICAL EMISSIONS FROM MODERN MUNICIPAL SOLID WASTE INCINERATORS IN ONTARIO

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Appendices

Appendix A-I	Detailed Methodology for Assessing Health Risks of MSW Incinerator Emissions.
Appendix A-II	Modelling of Emission Rates, Air Concentrations and Population Risk due to an Incinerator.
Appendix A-III	Distributions for Exposure Variables Used in MSWI Probabilistic Analysis.

Summary

In this study, a human health risk assessment was performed to evaluate emissions from a hypothetical MSW incinerator located in southern Ontario near a major urban centre.

The assessment showed that human health impacts were negligible based on the procedures used and health endpoints evaluated.

The overall conclusion is that the potential for adverse human health effects in populations residing near modern, well-run MSW incinerators which meet existing air emission criteria should not be a limiting factor in the selection and approval of these facilities.

A-1.0 INTRODUCTION

The ministry undertook this evaluation of the potential health risks associated with disposal of municipal solid waste (MSW) in modern municipal incineration facilities as a timely response to current public concerns about the disposal of MSW, and, the need to assess current scientific information on emissions from modern, well-run facilities. Accompanying reports evaluate potential health and ecological risks of MSW incinerators and MSW landfills (Report Parts B, C and D)

An important goal of this investigation was to provide interested parties with a comparative technical assessment of two waste disposal methods. For this purpose, this predictive study examined realistic scenarios regarding the technical aspects associated with state-of-the art facilities as they may be promoted in the province of Ontario, the nature of their emissions, the populations affected, and a spectrum of health effects that may be reasonably addressed within a predictive risk assessment framework.

In 1992, Ontario banned the construction of new incinerators for burning municipal waste (Ont. Reg. 555/92). Also, in 1992, under the Waste Management Act, MSW incineration was precluded as a disposal option in replacing waste disposal capacity for existing landfill sites in the GTA. Other jurisdictions in Canada and elsewhere continue to allow new waste incinerators to be built, and debate over the pros and cons of incineration vs. landfill has continued.

The Ministry has recently decided to lift the ban on new MSW incinerators. Following public comment on proposed legislative changes placed on the Environmental Bill of rights registry in late 1995, Guideline A-7 has replaced Guideline A-1. More stringent air emission limits have been placed on new MSW incinerators. These new air emission limits will also be considered during the expansion or modification of existing incinerators.

This assessment is based on emission characteristics of existing modern municipal incineration facilities. A new municipal waste incinerator would be required to meet Ontario's stricter Guideline A-7 requirements. As a result, it would be expected that a new incinerator would present even less risk than shown in this analysis.

There is a considerable literature on the health risks of MSW incinerator emissions (see Bibliography). This study builds on this body of information.

A-1.1 Approach

This study uses a risk assessment framework (NRC,1983; EPA,1986, CEPA, 1994) to evaluate the environmental and human health risks of MSW incinerator and MSW landfill air emissions. Risk assessment is defined as the scientific evaluation of the probability of adverse consequences (either direct or indirect), and the accompanying uncertainties, to the environment and to human health caused by physical, chemical or biological disturbances to the environment. In this case, the health risk assessment consists in estimating the potential carcinogenic and non-carcinogenic risks that may result when chemicals are released from municipal waste disposal facilities.

This study only addresses potential health risks in populations near hypothetical facilities. Technology and economic aspects are not addressed. Nor are the 3 Rs. Modern risk assessment paradigms and methodologies were used to assess receptor exposure to emissions and potential health or environmental effects associated with these exposures. The intent of the risk assessment was to provide an up-to-date and realistic estimate of the potential for adverse health effects in humans living near MSW incinerators for most of their lives.

A multitude of mathematical models exist to estimate the various components of the four major phases of risk assessment (hazard identification, exposure assessment, dose-response assessment and risk characterization). The human health risk assessment methodology used the most recent federal, Ontario and U.S. toxicological information available. Recent Ontario guidelines and policies pertaining to air quality were also incorporated where applicable (OMOEE, 1994; CEPA, 1994; IRIS, 1995; NATICH, 1992; HEAST, 1995; CAPCOA, 1993). Where possible, use of Ontario-specific information was maximized.

A-1.2 Limitations of the Approach

Modern risk assessment techniques, which borrow from disciplines such as toxicology, environmental sciences, and public health are extensively used today by regulatory agencies and health organizations in order to formulate research priorities, identify potential environmental health concerns, and provide a basis for the development of protective regulations. When used properly, these approaches allow for early recognition of potential health hazards and the circumstances under which these hazards may be expressed, thus permitting the development and implementation of preventive measures. Because of the limitations inherent to any predictive approach, however, certain possible outcomes can only be hypothesized and, consequently, must await further confirmation through field studies. As such, predictive risk assessments must be viewed as constituting a filtering process that helps formulate inferences within the limits of what can be predicted based on the available scientific evidence.

The use of deterministic point estimate (also "reasonable worst case" i.e. modified CAPCOA) risk assessment methodologies based on upper bound or otherwise conservative point estimates is well established in modern regulatory procedures (Birmingham, 1990, CanTox, 1987, 1988, CRC, 1991, Jones, 1994, Levin et al., 1991, Macpherson, 1987, CAPCOA, 1993, U.S. EPA, 1990, 1993). While providing conservative estimates of potential risks, the method has been challenged. A weakness of the approach is that the degree of variability and uncertainty in the risk estimate is not transparent. Some authors have claimed that risks are over-estimated (Copeland et al., 1994, Finley and Paustenbach, 1994). To combat these problems, a probabilistic approach which incorporates frequency distributions to describe ranges of exposure has evolved. When combined with toxicological limits, this probabilistic approach generates distributions of risk rather than point estimates and so provides more information to guide risk managers in regulatory decisions

(Copeland et al., 1994, Finley and Paustenbach, 1994, Cullen, 1995, Seigneur et al., 1995, Smith, 1994, Thompson et al., 1994). It should be noted that all risk assessment procedures contain some element of uncertainty. Consequently, the output or result of a risk assessment including a probabilistic one is an ESTIMATE of risk and not the actual risk.

This assessment attempted to examine the potential human health risks attributable to typical MSW incinerator emissions without considering the background levels of chemicals. Background levels are generally specific for a geographical location and, consequently, may vary considerably depending on the presence of natural and anthropogenic sources. Because the main objective of the present analysis was to investigate the potential baseline health risks of the facility, the background aspect was intentionally not considered.

Based on the above considerations, caution must be used to assure that the underlying assumptions behind the information provided from this facility are valid for any purposes or scenarios to which these results are utilized. Prime amongst these assumptions for this assessment are the state-of-the-art nature of the emission control devices at the generic incinerator (a new municipal waste incinerator would be required to meet Ontario's stricter Guideline A-7 requirements). There is also an assumption that the facility does not experience a significant failure of any pollution control system. Consequently, no failure scenarios were evaluated.

A-2.0 METHODOLOGY

A-2.1 Overview of hypothetical facility, site and emissions characterisitics.

In this study, a hypothetical MSW incinerator was co-located at a hypothetical landfill site in a southern Ontario location adjacent to a major urban center (see MSW landfill report) and many site characterisics relating to local meteorology, soil , groundwater, human receptors, etc., were shared. The MSW incinerator considered for this analysis was defined based on existing, modern technological and scientific criteria currently recommended and/or accepted by the Ontario Ministry of the Environment .

These criteria include, among others, considerations for the choice of an appropriate location based on variables such as a) projected capacity and operational lifespan of the facility, b) existing and projected human populations, c) natural ecosystems (terrestrial and aquatic), d) geomorphological and hydrogeological factors, and e) site meteorology (*e.g.*, dominant winds). This methodological choice was dictated strictly by practical considerations and does not bear any implication for the site(s) in the foreseeable future.

The location for the incinerator scenario is the proposed Durham landfill site EE11. The site is located in Southern Ontario in a populated area near the ecologically-important Rouge River. Details of the watershed, including surface and groundwater aspects, are included in reports of the Interim Waste Authority (IWA 1994a, 1994b).

Meteorological conditions at the site are based on data from Stouffville Airport (wind speed and direction and temperature) and Buttonville Airport (cloud cover) (IWA 1994a p4-4).

For the hypothetical population residing around the facilities, actual census tract data was used. For the purposes of this study, census data for the population (about 555,000) living around the Britannia Road landfill in Mississauga was used as a surrogate.

Monitoring data from the Peel Resource Recovery Inc. (PRRI) Incinerator (now called KMS Peel Inc.) (Air Testing Services, 1993, Bovar-Concord, 1995) and the Hartford Incinerator (Environment Canada/ U.S. EPA, 1991, 1994) were used. These facilities were selected as representing intermediate and large sized state-of-the-art facilities burning 400 tonne/day and 1500 tonne/day, respectively.

Incineration of MSW creates flue gases which are treated with air pollution control (APC) equipment to meet stack emission or POI limits. The main residue of the incineration process is the bottom ash which falls through the grating below the boiler and other unburnt material (siftings) caught on the grating. Flyash is the particulates removed from the flue gases by the APC equipment.

The total ash generated is about 25% by weight and 10% by volume of the MSW sent to the incinerator. The flyash component is about 10% of the total ash or 2% to 3% of the original waste.

Currently, bottom ash (and siftings) are classified as non-hazardous and are co-disposed with regular MSW in conventional landfill sites. Under Ontario Reg. 347, flyash must be handled seperately from bottom ash. Flyash must be tested to see for "leachate toxic" for it to be classified as non-hazardous. Depending on leachate quality, flyash that is hazardous is disposed of as hazardous waste at an approved "secure" disposal facility, while non-hazardous flyash goes to conventional landfill.

A-2.2 Human Health Risk Assessment

The MSW incinerator health risk assessment attempted to estimate potential health risks associated with atmospheric emissions of carcinogenic and non-carcinogenic toxicants through inhalation, ingestion of locally grown produce and dermal exposure to soil. A risk assessment of landfilled incinerator ash (mainly leachate related) could not be performed due to information and time limitations. A preliminary assessment of the potential health impact of MSW incinerator bottom ash disposal in landfill based on the available literature is presented (section A-4.0).

The risk assessment consisted of 3 main phases:

1) Prioritization of chemicals in MSW incinerator air emissions, and, selection of a critical list of

chemicals based on worst case inhalation exposure conditions,

2) Modelling of these critical chemicals into plausible multiple exposure pathways based on long term exposure to the maximum ground level concentration (GLC),

3) estimation of the intake of critical chemicals by human receptors and characterization of potential health risks.

To maintain comparibility with existing risk assessment methodologies, the MSW incinerator risk assessment initially used a deterministic approach based on point estimates of means or upper limits (CAPCOA, 1993). A probalistic analysis (Monte Carlo procedure) was then performed using the Crystal Ball ™ software (Decisioneering, 1994) to provide a more realistic analysis.

A-2.3 Initial Screening and Selection of Critical Chemicals

Incineration of waste leads to emissions of a large number of chemicals. However, resource and information constraints mean that not all of the emitted chemicals can be studied for their effects on human health. In phase one, the emitted chemicals which are the most abundant and/or have the most severe, adverse impact on health based on carcinogenic or non-carcinogenic effects were investigated further. The screening protocol used in this risk assessment to identify these critical compounds used monitoring data from the PRRI Incinerator and the Hartford Incinerator. Any compound which appeared on either monitoring list was considered for the initial stage of screening (in excess of 130 compounds were analysed for).

This initial listing was reduced by removing about 45 compounds which were not detected at either facility. Detection limits for analysis of Peel and Hartford incinerator emissions were in line with what is achievable by commercial analytical laboratories and generally met analytical detection requirements of MOEE and the US EPA. This does not mean that any of these chemicals were not present. Due to the large uncertainties involved in assessing what might be present below the detection limit, it was decided that the 45 compounds would not be considered for assessment. Another 14 compounds were measured in incinerator emissions for which no reliable information on their toxicology was found. This was the last stage in preparing the list of about 70 compounds to be considered for the screening stage of the risk assessment (Table A-1).

In the second phase of the screening, maximum human inhalation exposure potential was used to select for critical chemicals. The first step was to model dispersion of the stack emissions. Modelling the dispersion of emissions from point sources such as incinerators is well established. Sophisticated mathematical models for all aspects of the dispersion of emissions and their ensuing environmental fates in the air, soil, water, sediment and biological compartments of the environment are well described (see Bibliography).

A brief overview of the various facets of the environmental fate of incinerator emissions are shown in Figures A-1 and A-2. Figure A-1 is a diagram of the physical extent of the problem.

Typically, the incinerator stack emissions are released from a fixed point usually greater than 30 m above ground level. The emissions are transported downwind and dispersed depending on the local topography and the prevailing wind and weather conditions. Locally there will be a region or zone of maximum impact (this point of impingement (POI) is associated with maximum ground level concentration (GLCmax)) which can be delineated with hypothetical isopleths (for example, see Figures A-II-3 to A-II-8 in Appendix A-II). This zone will occur within kilometres of the source and may impact on urban sites, rural areas or streams and lakes.

Inside this zone of impact which can be delineated mathematically, a number of models can be used to evaluate the potential exposure of receptors. Figure A-2 gives a simple outline of the pertinent models and exposure pathways. As can be seen the potentially significant pathways to humans (examined in this study) are:

a) direct inhalation of dispersed emissions, and,

b) indirect pathways through ingestion of backyard produce that may have absorbed chemicals from the emissions. Direct dermal contact with soils or dusts contaminated by emissions is also possible.

Normally, site assessment for a MSW incinerator would involve a multi-media approach to exposure assessment with estimates of human exposure from all exposure pathways, i.e., through inhalation, ingestion of food, water and soil, and dermal exposure to soil, bathing water or consumer products (Birmingham, 1990; CanTox, 1987a,b, 1988;Levin etal., 1991; Macpherson, 1987; U.S. EPA, 1990, 1993). In the selection of critical chemicals, only the contribution of the MSW incinerator emissions to ground level air quality was considered.
Other exposures to soil contaminated by deposition of incinerator emissions or produce locally grown in this contaminated soil were not incorporated into the initial screening of existing MSW incinerators, but were used in the exposure assessment of the critical chemicals.

For this study, the dispersion model used was the MOEE Clean Air Program (CAP), Appendix 8 model. Modelling of short term and annual ground level concentrations (GLC) was done by STB staff (Appendix A-II). The stack emission data used for modelling POI/GLCmax were maxima from monitoring data when the facilities were operating at full capacity. This is conservative since operation of incinerators can involve up to 25% down time due to maintenance and repair.

In this phase of the screening, the maximally exposed individual (MEI) is an adult human receptor assumed to live in the zone near the incinerator where maximum GLC occur on a long term basis. The main focus for screening was on the calculated inhalation exposure of this receptor to 24 hr GLC maxima. Short term GLC maxima are based on daily weather patterns and exhibit greater variability and give much larger GLC values than maxima baed on annually averaged weather data. The modelling was performed using a meteorology data set typical of southern Ontario, but using worst case air bouyancy. Under these conditions, ground level concentrations are maximized. This over-estimate of inhalation exposure coupled with the use of long term health criteria is a worst case scenario suitable for screening chemicals with high intakes and/or health impacts.

In all phases of the screening and assessment, potential adverse health effects of the critical chemicals screened from the first screening list were characterized. Health effects may be non-carcinogenic, ie. adverse effects are observed dermally or in various internal organs or biochemical functions or may be effects on reproductive health and development, the immune system or the nervous system, or, effects may be carcinogenic.

For non-carcinogenic effects, reference doses (RfD) or reference concentrations (RfC) were used. Generally, this is the dose or concentration at which no adverse effects are expected. In most cases, these RfD (also called acceptable daily intakes (ADI) or tolerable daily intakes (TDI)) already incorporate "safety" factors to reduce the limit below minimum effect levels. In some instances (for example particulate emissions), health effects data were not available but regulatory guidelines were found and used in lieu of a RfD/RfC.

The hazard of non-carcinogenic chemicals was assessed using the exposure ratio (ER). The ER was calculated by dividing the calculated intake by acceptable exposure limits (RfC, TDI, RfD, etc.). An ER of 1 or less indicates a safe level of exposure; an ER of greater than 1 indicates that the receptor may be exposed to levels of the chemical which may cause adverse health effects.

Lifetime cancer risk estimates were calculated by multiplying calculated receptor intakes by the appropriate risk per intake factor. Carcinogenic effects were recorded as risk per unit exposure, since most carcinogens are considered to have non-threshold dose-response curves. The exceptions are dioxins and furans. This family of compounds was treated separately from the other carcinogens in that a risk factor is not involved and a tolerable daily intake (TDI) (similar to a RfD is used).

Selection of critical chemicals was based on the cumulative estimate of calculated cancer risk or ER of the 70 chemicals from the initial screening. Those chemicals contributing to 99% of the combined cancer risk or ER for the MEI were placed on the critical chemical list (Table A-2).

As well as modelling the carcinogenic or non-carcinogenic potential of individual chemicals, the chemicals were also grouped according to their major adverse effect so that the risk for these major endpoints could be aggregated.

A-2.4 Deterministic Assessment of Critical Chemicals

In the assessment of the critical chemicals (Table A-2), annual average GLC maxima were assumed to be inhaled on a lifetime basis (lifetime average daily dose or LADD). Intakes from contaminated soil or local produce exposures were estimated on the assumption that the incinerator emissions deposited on the soil for 70 years and were incorporated into the upper 5 cm, and, local produce grown in this soil was consumed. The soil and produce exposure intakes were calculated using equations in CAPCOA (1993) (Appendix A-I). Exposures to tapwater and normal food items were not considered. Exposures to tapwater, groundwater, farm produce (beef and milk) or supermarket produce, or local fish consumption pathways were not considered given the suburban lifestyle of the population near the facility. This lifestyle assumes that the major human intakes of water and food would be consumption of treated tapwater and supermarket food stuffs which would be obtained from sources not affected by the local incinerator or landfill. Under these circumstances, the breastmilk pathway for nursing infants would reflect the major dietary intakes of the mother and would constitute a non-significant pathway for MSW incineration emissions.

Only inhalation exposures were calculated for particulates, acid gases and airborne materials which only present a risk if inhaled (silicon, iron, phosphorus, tin).

In the case of cadmium and chromium, ingestion exposures were treated as non-carcinogenic, since carcinogenicity has not been demonstrated by this route of exposure for these metals.

For the critical chemicals selected, each of these exposures can be integrated to assess their potential risks.

Details of the deterministic risk assessment of the critical chemicals are described in Appendix A-I.

A-2.5 Probabilistic analysis of Critical Chemicals

The standard point estimate approach, as used in U.S. jurisdictions (CAPCOA, 1993; U.S. EPA, 1990, 1993), attempts to compensate for the uncertainties inherent in many exposure parameters by selecting worst case values. In this study, some of the CAPCOA defaults were modified to less stringent values based on a more realistic choice of values from the literature. This approach might be better described as a reasonable worst case one.

Probabilistic techniques (also referred to as Monte Carlo, Latin Hypercube, quantitative uncertainty or distributional analyses) offer a more refined approach by assigning probability density functions to exposure variables. This allows the risk assessor or manager to use the distribution of risk estimates for various exposure scenarios to make decisions on the level of risk associated with regulatory applications. Probabilistic techniques have been well described in recent years (Finley and Paustenbach, 1994; Smith, 1994; McKone and Bogen, 1991, Thompson et al., 1992, Finley et al., 1994). Software to run probabilistic analyses on desktop computers is also available (@Risk, Crystal Ball). The Monte Carlo approach has been applied to incinerator emissions (Copeland et al., 1994; Cullen, 1995). It should be noted that the output of Monte Carlo analyses also has its own degree of uncertainty related to selection of probability density functions for variables and chemicals lacking robust data. Monte Carlo analyses of the modified CAPCOA equations were performed using Crystal Ball® software (Decisioneering Inc., 1994).

Details of the Monte Carlo analyses are described in Appendices A-I and A-III.

A-2.6 Population risk of Critical Chemicals

Generally, population risks for communities located near incineration facilities have not been computed for most traditional incinerator risk assessments. Since the lifetime cancer risks are usually estimated for the MEI, this maximum individual risk cannot be extrapolated to the whole population since the majority are not maximally exposed. For actual communities, census tract data can used to calculate hypothetical cancer burdens (Park, 1990). For the purposes of this study, census data for the population (about 555,000) living around the Britannia Road landfill in Mississauga was used as a surrogate. The methodology for estimating population risk is based on placing a population grid based on this population around the MSW incinerator facility. Air concentrations in each grid cell based on dispersion from the MSW incinerator were used to assess population risk in that grid cell. Gridwide risk estimates were integrated to obtain an overall estimate of population risk expressed as the number of cancer deaths attributable to MSW incinerator emissions during the period of operation of the facility. Details of the methodology to estimate lifetime excess cancer risks (or the number of possible cases) in this surrogate population are detailed in Appendix A-II.

A-3.0 RESULTS

A-3.1 Screening of MSW incinerator emissions for critical chemicals and point estimate determination of risks to MEI.

The list of about 70 compounds to be considered for the screening stage of the risk assessment is shown in Table A-1. A critical list of chemicals was selected from the 70 compounds in Table A-1 using conservative inhalation risk modelling (Appendix A-1.3). The results of the first phase of the assessment to select a short list of chemicals that represents 99% of the potential for adverse health effects that might result from direct inhalation of MSW incinerator emissions is shown in Table A-2. These chemicals were studied further.

In the second part of the screening exercise, annual average GLC were used as inputs to the soil deposition/incorporation model. The estimated soil concentration resulting from long term incineration emissions was used to estimate soil ingestion and dermal exposure intakes. The long term soil concentration was also used to estimate uptake into back yard produce. Most of the assumptions, factors, intermediate calculations for these intake estimates are shown in Appendix A-1.4. Results of the inhalation, dermal and ingestion exposures to MSW incinerator emissions are shown in Table A-3

There are six chemicals with carcinogenic potential and nine which may result in non-carcinogenic effects (Table A-2). As well as modelling the carcinogenic or non-carcinogenic potential of individual chemicals, the chemicals were also grouped according to their major adverse effect so that the risk for these major endpoints could be aggregated (Table A-3). In this case, inhaled chromium and cadmium contibute to respiratory tract tumors, arsenic, vinyl chloride, and dioxins contribute to internal tumors, mainly liver tumors, and benzene mainly is associated with leukemia. For non-carcinogenic chemicals, the acidic gases (nitrogen dioxide, sulphur dioxide, hydrogen chloride) mainly act as pulmonary irritants. Particulate bound elements, such as silicon, iron and tin and the particulates themselves can provoke a number of pneumoconioses (inflammatory conditions leading to fibroses in the lung). Mercury and lead cause neurotoxic effects.

Since the acidic gases, particulates and silicon, iron and tin only pose a health hazard following inhalation, other pathways of exposure were not examined further. All other critical chemicals were evaluated for indirect soil and back yard produce exposures as well as inhalation exposures.

The results of the point estimate risk assessment of critical chemicals are shown in Table A-4. Lifetime inhalation of five critical contaminants, which have carcinogenic potential was estimated to have a conservatively calculated lifetime risk from inhalation of 8.1×10^{-8}. Lifetime exposure to soil exposure and produce consumption due to these five carcinogens in municipal waste incinerator emissions was estimated to have a much lower risk than inhalation. For these five carcinogens, inhalation risk is the major risk contributor for the volatile organics, e.g., >99% for benzene and about 86% for vinyl chloride. By contrast, the inhalation risk for arsenic is 44% of the total risk.

For dioxins, less than 0.03% of the ER for all exposure routes is due to inhalation.

The other 9 compounds with non-carcinogenic effects in these municipal waste incinerator emissions were estimated to have a cumulative ER of 0.1. This cumulative ER includes the non-carcinogenic ER due to non-inhalation pathways of chromium and cadmium.

For gaseous irritants and particle bound pneumoconiotic agents, the major hazard is via inhalation. For lead and mercury, indirect exposure pathways of exposure contributed 60% and 86% of their overall ER, respectively. This result and the indirect exposure result for dioxins indicates the importance of multi-media exposure assessment when assessing atmospheric emissions.

Although dioxins are considered to have some potential to be human carcinogens, their biological mode of action suggests that a threshold probably exists for this effect. Evidence from studies of the non-carcinogenic effects of dioxins, particularly reproductive toxicity suggests that this non-carcinogenic effect may be as important as any carcinogenic potential. For these reasons, the toxicity of dioxins was evaluated using the ER approach in conjunction with a very stringent TDI (CEPA, 1990).

The odour potential of MSW incinerator emissions was not assessed, thus, no conclusion as to odour impacts can be made.

A-3.2 Probabilistic analysis of risks of MSW incinerator emissions to MEI.

The results of the probabilistic analysis are shown in Tables A-5 and A-6. For clarity, results for contributing to particular endpoints have been combined and only the median, 95th and 97th percentile risks are shown. For comparison, the results of section A-3.1 are also included.

A probabilistic analysis of the risk distribution for the MEI showed that the upper risk distribution (median to 95th percentile) for combined carcinogenic risks ranged from 4.7 X 10^{-8} to 2.3 X 10^{-7}. For non-carcinogenic chemicals, the combined ER estimate ranged from a median value of 0.06 to 0.33 at the 95th percentile. The ER for dioxins alone was estimated to range from 0.06 (median) to 0.15 (95th percentile).

The upper range of the distribution of risks (median to 95th percentile) of the MEI estimated using the Monte Carlo procedure overlap the point estimate values derived in section A-3.2.1. In most cases, the point estimate risk value is less than the 95th percentile. This is a surprising result since the literature suggests that point estimate procedures usually overestimate risk. The Monte Carlo results indicate that the point estimate methodology used was not in fact a worst case estimate.

A-3.3 Probabilistic analysis of population risks to MSW incinerator emissions.

Population cancer risk estimates were based on placing the population around the Britannia Road landfill around the MSW incinerator facility. Integrated cancer risk estimates for this population indicated that there were no cancer deaths attributable to MSW incinerator emissions during the period of operation of the facility (Table A-7).

At this time, ER estimates could not be used to estimate the incidence of adverse effects in a large population. However, if the relationship between the predicted cancer risks for the MEI and the population around the facility is similar for ER, then adverse effects for the population at large are unlikely.

A-4.0 ASSESSING THE RISKS FROM MSW INCINERATOR ASH DISPOSAL.

A-4.1. Current Situation.

Originally, the main reason for MSW incineration was to reduce the quantity and volume of waste to be disposed of. Energy recovery, as steam heat or electricity is now an important component of waste incineration. Incineration of MSW creates flue gases which are treated with APC equipment to meet stack emission or POI limits. The main residue of the incineration process is the bottom ash which falls through the grating below the boiler and other unburnt material (siftings) caught on the grating. Flyash is the particulates removed from the flue gases by the APC equipment.

The total ash generated is about 25% by weight and 10% by volume of the MSW sent to the incinerator. The flyash component is about 10% of the total ash or 2% to 3% of the original waste.

Currently, bottom ash (and siftings) are classified as non-hazardous and are co-disposed with regular MSW in conventional landfill sites. Under Ontario Reg. 347, flyash must be handled seperately from bottom ash. Flyash must be tested and pass "leachate toxic" criteria for it to be classified as non-hazardous. Depending on leachate quality, flyash that is hazardous is disposed of as hazardous waste at an approved "secure" disposal facility, while non-hazardous flyash goes to conventional landfill.

Current CCME ash management guidelines (CCME, 1989, Sawell and Constable, 1993) recommend that:

- ashes from all new MSW incinerators should be characterized by testing of the different ash streams.

- bottom ash and flyash streams be seperated and that flyash should not be mixed with MSW.

- while co-disposal of bottom ash with MSW in conventional landfill with appropriate liners, etc., is permitted, CCME is examining the option of monofilling bottom ash. Ash monofills have the potential of providing a stable alkaline pH which would limit metal solubility in leachate.

At the present time, no information on MSW incinerator bottom ash monofills could be located. According to other estimates (Chrostowski and Sager, 1991), about one third of U.S. MSW incinerator ash is disposed of in monofill sites. However, U.S. legislation permits the mixing of flyash with bottom ash for landfill disposal. Consequently, much of these data can not be used directly for the Ontario situation.

A-4.2. What is in Current MSW Incinerator Ash?

Most of our information is based on reports of regulatory testing of ash and leachate (Chrowstowski and Sager, 1991; Environment Canada/ U.S. EPA, 1994; Sawell and Constable, 1993). Flyash characterization is not discussed further because most flyash from modern MSW incinerators in Ontario is disposed of as hazardous waste by a number of processes including solidification and disposal in secure landfills.

Regulatory testing of bottom ash is limited to physical and chemical properties, selected inorganics, metals and semivolatile organics (listed in table 8). No information on the presence or content of other more volatile organic compounds such as BTEX or similar (tested in stack emissions) in bottom ash was located.

Recent reviews of field testing of leachates from landfills receiving only MSW (Kerry Rowe, 1994), landfills where incinerator ash of various types has been co-disposed with MSW (Chrowsowski and Sager, 1991, Ruland et al., 1994) and combined bottom ash and flyash monofills (Chrowsowski and Sager, 1991) indicate that a much wider spectrum of inorganic and organic chemicals have been found in leachate. As illustrated in Table A-8, many of these leachate constituents exceed drinking water guidelines.

A-4.3 Estimating Exposure

Modern landfill design for MSW or MSW ash disposal includes a clay liner usually in conjunction with a synthetic liner and leachate collection system. The purpose of any liner system is to retard the movement of leachate from the landfill to the subsurface groundwater. The 1998 Landfill Standard (Schedule 3) assumes that the primary synthetic liner has a service life of 150 years. Clay liners exposed to dilute aqueous leachate are unlikely to degrade. Clearly, unless the integrity of the synthetic and clay liner system is breached, the leachate collection system will divert the leachate to the leachate treatment system and seepage through the liner systems should be minimized. Similarly, the health risk to an off-site human receptor drinking groundwater from a well should be negligible (IWA, 1994).

To assess the health effects of gradual seepage of leachate components or some sort of failure of the landfill liner/ collection/ leachate treatment system, the literature on the fate of leachate "plumes" from existing landfill sites can indicate the potential for migration of bottom ash contaminants to groundwater. Recent assessments of the mobility of organic and inorganic chemicals in leachate (Kerry Rowe, 1994, IWA Detailed Site Assessment - Appendix QD, 1994, Ruland et al., 1994) indicate that many physical, chemical and biological processes in the unsaturated and saturated (below water table) groundwater zone under the liner can reduce the leachate concentration to or below acceptable levels at some distance from the landfill. This reduction in concentration is called attenuation.

The main processes of subsurface attenuation include dilution, adsorption, biodegradation, chemical exchange reactions, chemical precipitation and filtration. Critical contaminants in leachate are characterized by the following properties: toxic potency, mobility, and, persistence in the local groundwater system. Based on testing of MSW incinerator bottom ash, chemicals of concern include heavy metals, dioxins/furans, PAH and chlorophenols. Toxic trace metals (cadmium, chromium, lead and mercury) and other metals such as iron, copper, manganese, zinc, are generally not mobile at the neutral pH levels found in southern Ontario groundwaters. Anaerobic groundwater conditions near landfills can permit some metal mobility. Kerry Rowe (1994) cites the work of Yanful et al.(1988) at the Sarnia landfill to suggest that heavy metals (cadmium, chromium, lead, mercury) and iron in leachate are not critical because they are essentially immobile in the clay liner and the clayey soils with $pH > 7$ and high carbonate content common in southern Ontario. Persistent organics such as the chlorinated dioxins/furans and the high molecular wt PAHs tend to bind to organic matter in soil and are noted for a lack of soil mobility in the absence of solvents or oils to transport them. Chlorinated phenols can migrate through groundwater, however, the levels found in leachate are well below drinking water criteria and further attenuation renders them insignificant. In field studies of landfill leachate plumes, metals typically migrate less than a meter from the landfill. In soil profile studies, dioxins only move cm distances over a decade (Rordorf et al., 1986)).

A-4.4 Identifying Health Effects

Due to information and time constraints, groundwater migration of MSW bottom ash chemicals could not be modelled for risk assessment purposes. The landfill leachate literature suggests that the more toxic components (heavy metals, dioxins, PAHs) are unlikely to migrate from a modern landfill even if the liner fails. Without comprehensive testing of bottom ash and its associated leachates, one can not preclude other chemicals from having some impact on human health, even though the potential health risks from known constituents of bottom ash appear to be negligible. It is recommended that future assessments of MSW incineration model human exposure to ash leachate more closely.

A-5.0 DISCUSSION AND CONCLUSIONS

In this study, a risk assessment was performed to evaluate potential health effects of emissions from a hypothetical MSW incinerator located in a southern Ontario location adjacent to a major urban center.

The risk assessment consisted of 3 main phases:

1) Prioritization of chemicals in MSW incinerator air emissions, and, selection of a critical list of chemicals based on worst case inhalation exposure conditions,

2) Modelling of these critical chemicals into plausible multiple exposure pathways based on long term exposure to the maximum GLC,

3) estimation of the intake of critical chemicals by human receptors and characterization of potential health risks.

To maintain comparibility with existing risk assessment methodologies, the MSW incinerator risk assessment initially used a deterministic approach based on point estimates of means or upper limits. A probalistic analysis was then performed to provide a more realistic analysis.

The risk or potential for adverse effects was evaluated from the exposure ratio (ER) for non-carcinogens or the incremental lifetime cancer risk for carcinogens. For carcinogens, calculated lifetime risks of one-in-a-million (10^{-6}) are considered negligible. Some jurisdictions also consider one-in-100,000 (10^{-5}) to be negligible or non-significant. An ER of less than 1.0 would also indicate that no adverse effects are expected.

The initial large list of chemicals (70) measured in MSW incinerator emissions (Table A-1) was screened down to a small list of chemicals (15) that accounted for 99% of the risk (Table A-2). Three risk modelling approaches were then used to estimate health risk. 1) a traditional reasonably conservative point estimate of risk to a MEI; 2) a probabilistic analysis of this MEI, and 3) a population-based estimate of risk to the community living around the MSW incinerator.

Following screening, maximum annual average ground level concentrations in air generated using emission and dispersion factors from a representative incinerator were used as inputs to the soil deposition/incorporation model. The estimated soil concentration resulting from long term incineration emissions was used to estimate soil ingestion and dermal exposure intakes. The long term soil concentration was also used to estimate uptake into back yard produce.

The results of the point estimate risk assessment of critical chemicals for the MEI are as follows. Lifetime inhalation of the five critical contaminants, which have carcinogenic potential, was estimated to have a conservatively calculated lifetime risk from inhalation of 8.1×10^{-8}. Lifetime exposure to soil exposure and produce consumption due to these 5 carcinogens in municipal waste incinerator emissions was estimated to have a much lower risk than inhalation. The combined inhalation, dermal and ingestion point estimate risk was about 8.3×10^{-8}.

A probabilistic analysis of the risk distribution for the MEI showed that the upper risk distribution (median to 95th percentile) for combined carcinogenic risks ranged from 4.7×10^{-8} to 2.3×10^{-7}. The 95th percentile cancer risk estimate for all carcinogenic exposures of 2.3×10^{-7} is considered to be "essentially negligible".

The nine compounds with non-carcinogenic effects in these municipal waste incinerator emissions were estimated to have a cumulative ER of 0.1 using the point estimate approach.

For thses non-carcinogenic chemicals, the Monte Carlo analysis yielded a combined ER estimate ranging from a median value of 0.06 to 0.33 at the 95th percentile. The 95th percentile ER for all non-carcinogenic exposures of 0.33 is well below the "safe" exposure limits proposed for these compounds.

Although dioxins are considered to have some potential to be human carcinogens, their biological mode of action suggests that a threshold probably exists for this effect. Evidence from studies of the non-carcinogenic effects of dioxins, particularly reproductive toxicity suggests that this non-carcinogenic effect may be as important as any carcinogenic potential. For these reasons, the toxicity of dioxins was evaluated using the ER approach in conjunction with a very stringent exposure limit. The point estimate ER is about 0.04 for the combined exposures to dioxins in MSW incinerator emissions. The ER for dioxins alone, using the probabilistic analysis, was estimated to range from 0.06 (median) to 0.15 (95th percentile). The 95th percentile of 0.15 for all dioxin exposures due to MSW incinerator emissions indicates a small contribution of these compounds to everyday exposures.

This study used conventional point estimate and probabilistic approaches to estimating exposure. It is notable that in this study, the upper range of the distribution of risks (median to 95th percentile) of the MEI estimated using the Monte Carlo procedure overlap the point estimate values. In most cases, the point estimate risk value is less than the 95th percentile. The Monte Carlo results indicate that the point estimate methodology used was not in fact a worst case estimate.

Population cancer risk estimates were based on placing the population around the Britannia Road landfill around the MSW incinerator facility. Integrated cancer risk estimates for this population indicated that there were no cancer deaths attributable to MSW incinerator emissions during the period of operation of the facility.

Groundwater migration of MSW bottom ash chemicals could not be modelled for this risk assessment. The landfill leachate literature suggests that the more toxic components (heavy metals, dioxins, PAHs) are unlikely to migrate from a modern landfill even if the liner fails. Without comprehensive testing of bottom ash and its associated leachates, one can not preclude other chemicals from having some impact on human health, even though the potential health risks from known constituents of bottom ash appear to be negligible.

The risk assessment used current exposure and toxicology data to screen for respiratory problems and several cancer endpoints in a MEI and the general population around a hypothetical MSW incinerator in southern Ontario. Using these methodologies and recognized criteria to evaluate the potential for health effects (lifetime cancer risk less than 10^{-5} to 10^{-6} or ER less than 1), no indications of health effects for the health endpoints examined were found. The study design was limited and health impacts from MSW incinerator ash disposed in landfill was not assessed directly. However, the majority of MSW incinerator risk assessments published in the literature do not predict adverse health effects, and the present study is consistent with this body of information.

CONCLUSIONS

The assessments showed that human health impacts were negligible based on the procedures used and health endpoints evaluated. The main findings are:

- The total combined carcinogenic risk from long term exposure to incinerator air emissions using a point estimate procedure was 8.1×10^{-8}. Estimated exposures from non-carcinogenic toxicants emitted were well below any exposure limits of concern.

- A probabilistic analysis of the risk distribution for the MEI showed that the 95th percentile for combined carcinogenic risks was estimated at 2.3×10^{-7}. For non-carcinogenic chemicals, the combined ER at the 95th percentile was estimated to be 0.33. The ER for dioxins alone was estimated at 0.15.

- Health risk estimates to the population surrounding the facility indicate that even one cancer death resulting from exposure to facility emissions was highly unlikely.

The overall conclusion of this study is that the potential for adverse human health effects in populations residing near modern, well run MSW incinerator which meet existing air emission criteria should not be a limiting factor in the selection and approval of these facilities.

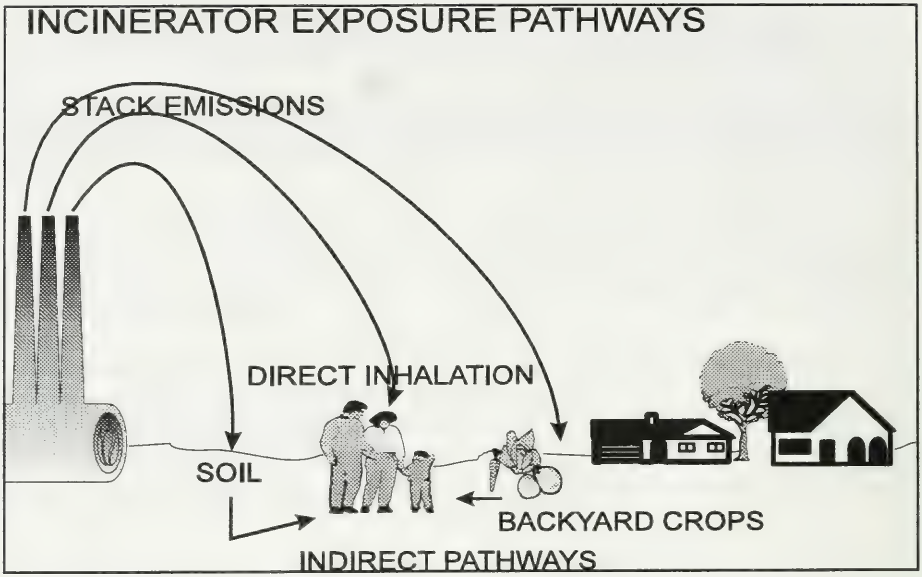

Figure 2. MSW Incinerator Exposure Pathways.

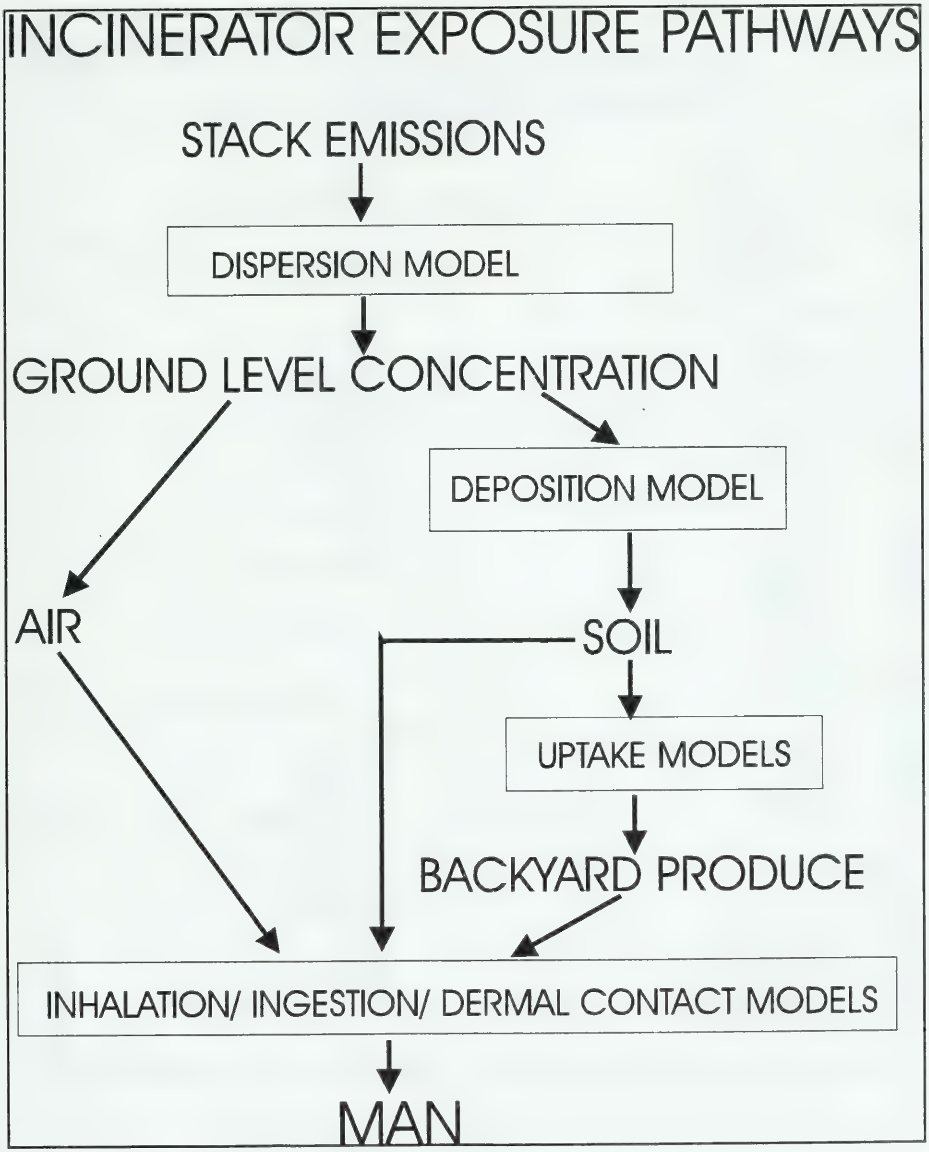
INCINERATOR EXPOSURE PATHWAYS
STACK EMISSIONS
DISPERSION MODEL
GROUND LEVEL CONCENTRATION
DEPOSITION MODEL
AIR
SOIL
UPTAKE MODELS
BACKYARD PRODUCE
INHALATION/ INGESTION/ DERMAL CONTACT MODELS
MAN

Figure 3. MSW Incinerator Exposure Models.

Table A-1 List of Chemicals detected in MSW Incinerator Emissions

Stack gases

- Particulate emissions
- Oxygen
- Carbon Dioxide
- Sulphur Dioxide
- Nitrogen Oxides
- Hydrogen Chloride
- Total Hydrocarbons (as CH_4)

Metal emissions

- Aluminum
- Antimony
- Arsenic
- Barium
- Beryllium
- Boron
- Cadmium
- Calcium
- Chromium
- Cobalt
- Copper
- Iron
- Lead
- Lithium
- Magnesium
- Manganese
- Mercury
- Molybdenum
- Nickel
- Phosphorus
- Selenium
- Silicon
- Silver
- Sodium
- Strontium
- Tin
- Titanium
- Vanadium
- Zinc
- Fluoride

Organics

Chlorobenzenes

- 1,2-Dichlorobenzene
- 1,3-Dichlorobenzene
- 1,4-Dichlorobenzene
- Total Trichlorobenzenes
- Total Tetrachlorobenzenes
- Pentachlorobenzene
- Hexachlorobenzene

Chlorophenols

- Total Dichlorophenols
- Total Trichlorophenols
- 2,3,4,6-Tetrachlorophenol

- Pentachlorophenol

Polycyclic Aromatic Hydrocarbons

- Tetralin
- Biphenyl
- O-Terphenyl
- 2-Methylnaphthalene
- 1-Methylnaphthalene
- Benzo(a)fluorene
- Benzo(b)fluorene
- Naphthalene
- Acenaphthylene
- Acenaphthene
- Fluorene
- Phenanthrene
- Fluoranthene
- Pyrene
- Benzo(b)fluoranthene

PCBs

- Total Chlorobiphenyls

Dioxins and Furans

- Total Dioxins and Furans (as I-TEQ)

Volatile Organics

- Chloromethane
- Bromomethane
- Dibromoethane
- Dichlorodifluoromethane
- Trifluoromethane
- Vinyl chloride
- 1,1-Dichloroethene
- Trichloroethene
- 1,1,1-Trichloroethane
- Tetrachloroethene
- Benzene
- Toluene
- Ethylbenzene
- Total Xylenes

Table A-2 Chemicals Assessed in Detail

Carcinogens	**Non-carcinogenic Toxicants**
Chromium	Nitrogen Oxides
Benzene	Silicon
Cadmium	Hydrogen Chloride
Vinyl Chloride	Mercury
Arsenic	Sulphur Dioxide
Dioxins	Iron
	Tin
	Particulate emissions
	Lead

Table A-3 Summary of lifetime exposures to MSW incinerator emissions

Chemical	LADD (ug/kg/d)				
	Inhalation	Soil Dermal absorption	Soil Ingestion	Food Ingestion	Total Ingestion + Dermal
Respiratory tract carcinogens					
Chromium	1.91E-06	1.07E-7	9.08E-7	8.14E-06	9.16E-06
Cadmium	9.14E-07	1.02E-8	4.34E-7	1.93E-05	1.98E-05
Internal (mainly liver) carcinogens					
Arsenic	2.80E-7	4.47E-10	3.79E-8	3.75E-7	4.14E-07
Vinyl Chloride	5.15E-06	1.31E-8	4.07E-8	5.68E-12	5.38E-8
Dioxins	1.04E-10	1.72E-12	3.39E-12	2.07E-10	2.12E-10
Leukemia					
Benzene	3.19E-04	7.51E-8	2.33E-7	2.26E-8	3.30E-7
Neurotoxicants					
Mercury	5.67E-04	5.55E-6	5.01E-05	2.85E-03	2.91E-03
Lead	3.05E-05	4.87E-8	4.14E-06	4.18E-05	4.60E-05
Pulmonary Irritants					
NO_2	4.72E-00				
SO_2	4.72E-02				
HCl	3.55E-02				
Pneumonicotic agents					
Silicon	5.33E-02				
Iron	2.89E-04				
Tin	2.17E-05				
Particulates	5.29E-03				

Table A-4 Summary of inhalation and total non-inhalation risks (point estimate) following lifetime exposures to MSW incinerator emissions

Chemical	Inhalation		Total dermal + ingestion		Total Inhalation + Non-inhalation	
	Risk	ER	Risk	ER	Risk	ER
Respiratory tract carcinogens						
Chromium	7.84E-8			9.16E-9	7.84E-8	9.16E-9
Cadmium	6.92E-10			1.97E-05	6.92E-10	1.97E-05
Internal (mainly liver) carcinogens						
Arsenic	9.88E-10		1.45E-9		2.45E-9	
Vinyl Chloride	1.81E-10		2.83E-11		2.10E-10	
Dioxins		1.04E-05		3.75E-02		3.75E-02
Neurotoxicants						
Mercury		4.50E-03		2.30E-02		2.76E-02
Lead		1.65E-05		2.48E-05		4.14E-05
Leukaemia						
Benzene	1.12E-9		1.15E-12		1.12E-9	
Pulmonary Irritants						
NO_2		4.07E-02				4.07E-02
SO_2		1.40E-03				1.40E-03
HCl		1.21E-02				1.21E-02
Pneumoconiotic agents						
Silicon		2.54E-02				2.54E-02
Iron		6.88E-04				6.88E-04
Tin		5.17E-04				5.17E-04
Particulates		2.52E-04				2.52E-04

Table A-5: Comparison of Deterministic and Probabilistic Estimates for Carcinogenic Risks to Individuals

Compound	Risk			
	Deterministic	Probabilistic		
		Median	95th percentile	97th percentile
Benzene (total risk)	1.12E-09	4.52 E-10	3.00 E-09	5.00 E-09
Chromium (inhalation risk)	7.84 E-08	4.31 E-8	2.15 E-07	3.61 E-7
Cadmium (inhalation risk)	6.92 E-10	5.29 E-10	1.86 E-09	3.19 E-09
Vinyl chloride (inhalation risk)	1.81 E-10	4.39 E-11	4.77 E-10	7.77 E-10
Vinyl chloride (indirect risk)	2.80 E-11	6.10 E-11	5.40 E-10	8.68 E-10
Arsenic (inhalation risk)	9.88 E-10	1.89 E-10	2.69 E-09	4.38 E-09
Arsenic (indirect risk)	2.45 E-09	2.28 E-09	7.93 E-09	1.41 E-08
Dioxins (total risk) - as an ER	3.75 E-02	5.58 E-02	1.46 E-01	2.58 E-01

Table A-6: Comparison of Deterministic and Probabilistic Estimates for Non-Carcinogenic Exposure Ratios

Compound	ER			
	Deterministic	Probabilistic		
		Median	95th percentile	97th percentile
Acidic Gases (total)	5.42 E-02	2.99 E-02	1.36 E-01	2.11 E-01
Cadmium (indirect)	1.97 E-10	1.12 E-5	8.38 E-05	1.51 E-04
Chromium (indirect)	9.16 E-09	8.76 E-09	3.11 E-08	5.85 E-08
Neurotoxics (total)	4.14 E-02	8.30 E-04	1.21 E-02	1.97 E-2
Pneumoconiotic Metals and Particulates (total)	2.68 E-02	6.40 E-03	7.03 E-02	1.18 E-01

Table A-7: Probabilistic Estimates of Carcinogenic Risks for the Population residing near the MSW incinerator.

Compound	Number of cases/ 550,000 persons		
	Median	95th percentile	97th percentile
Benzene (total risk)	3.90 E-08	5.48 E-07	8.96 E-07
Chromium (inhalation risk)	3.47 E-08	2.15 E-07	3.54 E-07
Cadmium (inhalation risk)	8.61 E-11	1.02 E-09	1.69 E-09
Vinyl chloride (total risk)	1.77 E-10	1.66 E-09	2.86 E-09
Arsenic (total risk)	9.11 E-11	1.25 E-09	2.14 E-09

Table A-8. Regulatory testing of contaminant concentrations (mean of 5 tests) in MSW incinerator bottom ash and leachate (Sawell et al., 1991).

Element/Compound	Bottom Ash	Reg. 309 Leachate	ODWO
Inorganic	mg/kg	mg/L	mg/L
Aluminium	49800	18.6	0.1 (AO)
Antimony	107	1.3	--
Arsenic	20.2	<0.006	0.025 (IMAC)
Barium	1088	0.43	1
Boron	165	1.94	5 (IMAC)
Cadmium	4.7	0.13	0.005
Chromium	1140	0.15	0.05
Cobalt	123	0.1	--
Copper	4276	154	1 (AO)
Lead	3230	8.17	0.01
Manganese	1136	6.42	0.05 (AO)
Mercury	1.69	0.0002	0.001
Nickel	998	1.1	--
Selenium	<0.5	<0.002	0.01
Silver	13	<0.1	--
Zinc	2042	60.8	5
Organic	ug/kg	ug/L	ug/L
Total PCDD+PCDF	0.2	0.1	0.000015
PAH	68.2	0.1	0.01
PCB	ND	ND	3
Chlorophenols	10.2	0.1	5 - 100
Chlorobenzenes	ND	ND	80 - 200

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APPENDIX A-I Detailed Methodology for Assessing the Health Risks of MSW Incinerator Emissions.

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A-I-1.0 Outline of Assessment Process

There are two main objectives of the assessment in this Appendix. The first objective is to assess what (if any) adverse health effects may be experienced by a community near a municipal solid waste incinerator (MSWI). The second objective is to quantify the potential for adverse health effects that are experienced by that community.

These objectives were achieved by using a five-step assessment process. The steps of the process and the question(s) that each step of the assessment was designed to answer are listed below.

1 **Facility and Site selection** - what are the characteristics and chemical emissions of the incinerator. Where is it located and what are the local geography, weather and human population characterisitics?
2. **Pre-screening** - what chemicals are likely to be emitted from a MSW incinerator?
3. **Screening** - which chemicals are of most concern because they cause the most serious adverse effects or are most abundant?
4. **Exposure assessment and Risk Characterization** - how much of each chemical emitted is calculated to reach the receptor and what is the estimated magnitude of the adverse effect?
    a. using the CAPCOA model
    b. using Monte Carlo simulations
5. **Population based risk estimates** - based on an hypothetical population around the incinerator site, how many cases of cancer would be expected due to normal operation?

Each of these steps is described in more detail within this appendix. The above outline is provided to guide the reader through the various steps of the assessment process.

A-I-1.2 Facility and Site Selection

The MSWI considered for this analysis was defined based on existing, modern technological and scientific criteria currently recommended and/or accepted by the Ontario Ministry of the Environment. Monitoring data from the Peel Resource Recovery Inc. (PRRI) Incinerator (now called KMS Peel Inc.) and the Hartford Incinerator were used (Bovar-Concord Environmental, 1995; Environment Canada / U.S. Environmental Protection Agency, 1991). These facilities were selected as representing intermediate and large sized state-of-the-art facilities burning 400 tonne/day and 1500 tonne/day, respectively.

A hypothetical MSW incinerator was co-located at a hypothetical landfill site in a southern Ontario location adjacent to a major urban center. Many site characterisics relating to local meteorology, soil , groundwater, human receptors, etc., were shared (the landfill site is described in detail in the accompanying report (Report B - *Predictive Assessment of the Potential Health Risks Associated with Chemical Emissions from Municipal Solid Waste Landfills*).

A-I-1.3 Characterization and Pre-screening of MSW Incinerator emissions.

The screening protocol used in this risk assessment used monitoring data from the PRRI Incinerator and the Hartford Incinerator. Any compound which appeared on either monitoring list was considered for the initial stage of screening (in excess of 130 compounds were analysed for).

This initial listing was reduced by removing about 45 compounds which were analysed for but not detected at either facility. Another 14 compounds were measured in incinerator emissions for which no reliable information on their toxicology was found. This was the last stage in preparing the list of about 70 compounds to be considered for the screening stage of the risk assessment (Table A-II-1).

A-I-1.4 Screening to select a critical list of chemicals based on worst case inhalation exposure conditions.

To screen the list developed in A-I-1.3, a worst case inhalation exposure scenario was used. Dispersion factors were used to estimate the maximum 24 hr GLC at the maximum POI from MSWI stack emissions (A-I-1.4.1). The calculated inhalation exposure for a maximum exposed individual (MEI) at this location was used to estimate risk based on lifetime exposure to maximum 24 hr GLC emissions (A-I-1.4.2). Potential for health risk was estimated (A-I-1.4.3). The risks or exposure ratios for all chemicals were summed and the cumulative risk or ER for individual chemicals was calculated as a percentage of the total risk or ER (A-I-1.4.4). The list of critical chemicals is based on the chemicals contributing to 99% of the total risk or ER. A limited number of chemicals that are of concern to the public were added even though the risk or ER was less than 1% of the overall risk.

A-I-1.4.1 Determining maximum 24 hr GLC

GLC maxima and the location of POI for both the Peel and Hartford facilities were modelled by STB staff using OMOEE CAP air dispersion models as described in Appendix II (Table A-II-2). The GLC dispersion factors and POI locations modelled using meteorology data typical of the southern Ontario location used (Buttonville/ Toronto International Airport) but with worst case air bouyancy. Under these conditions, GLC are maximized.

Modelling the maximum POI and GLC for both the Peel and Hartford incinerators using meteorological conditions typical of the southern Ontario site and worst case bouyancy conditions indicated that the dispersion factors for POI/GLC derived from the Peel facility gave the most conservative estimates. These Peel dispersion factors were used to estimate inhalation exposures of the MEI at the Maximum POI/GLC. GLC generated using Peel dispersion factors were used as inputs to the CAPCOA model.

A-I-1.4.2 Estimating Inhalation Exposure of MEI

Calculated Receptor Intake The calculated receptor intake (CRI) was calculated using the POI, a breathing rate of 23 m^3/day and 70 kg average body weight (IWA, 1994; CEPA, 1994).

$$CRI=23*POI/70 \text{ (units are } \mu g/kg/day).$$

This is the intake expected if a receptor is located at the POI.

A-I-1.4.3 Characterization of Potential Health Risk of MEI

Potential health risks for the MEI receptor associated with exposure to MSWI emissions were characterized by comparing health protection criteria identified in A-I-1.4.3.1 with inhalation intake estimates (A-I-1.4.3.2).

A-I-1.4.3.1 Databases Used for Identifying Health Effects

In order to choose the information for health effects and identify health protection criteria, literature was searched in the following order:

1. Ontario Ministry of the Environment

   RfD/RfC and carcinogenic risk estimates. These documents have been released to the public domain and have received external review.

2. Health Canada/Environment Canada

   Tolerable Daily Intakes (TDI) or risk estimates developed under the Priority Substances Evaluation Program, *Canadian Environmental Protection Act* (CEPA). These documents have been released to the public domain and have received external review.

3. US Environmental Protection Agency - Integrated Risk Information System (IRIS)

   IRIS provides peer-reviewed and agency supported values for RfCs, RfDs and carcinogenic risk estimates.

4. Other EPA and State Criteria

   National Air Toxics Information Clearinghouse (US EPA)(NATICH), Health Effects Assessment Summary Tables (US EPA) (HEAST), California Air Pollution Control Officers Association (CAPCOA) - these data have undergone peer review. HEAST and

NATICH may contain guidelines for chemicals which have been withdrawn from IRIS.

Toxicological assessments developed in Ontario or Canada were given preference because they are believed to be the most relevant to the scenario that is being studied. One exception to choosing Canadian data over that of other agencies was the case of hexachlorobenzene. The CEPA document did not provide sufficient information to calculate the unit risk of hexachlorobenzene; thus, the data from IRIS was used.

A-I-1.4.3.2 Estimation of Potential Health Risk of MEI

Inhalation Risk For carcinogens, the inhalation risk was calculated in two ways. If the risk estimate was given as risk per $\mu g/m^3$, then the following calculation was used:

$$\text{Inhalation risk} = \text{GLC*risk estimate}$$

If, however, the risk estimate was given as risk per $\mu g/kg/day$, then the following calculation was used:

$$\text{Inhalation risk} = \text{GLC} * \text{risk estimate} * 23/70,$$

where 23 represents a breathing rate of 23 m^3/day and 70 represents an average body weight of 70 kg (CEPA, 1994).

Exposure Ratio For non-carcinogens, the exposure ratio (ER) was calculated as follows:

$$\text{ER}=\text{GLC/RfC}.$$

An ER of 1 or less indicates a safe level of exposure; an ER of greater than 1 indicates that the receptor may be exposed to levels of the chemical which may cause adverse health effects. In order to be conservative in estimating the health impact of a mixture of chemicals, all chemicals with an ER of 0.1 or higher will be included in the detailed risk assessment.

A-I-1.4.3.4 Screening for Critical Chemicals in MSWI Emissions

Carcinogens - The final list of carcinogens to be assessed in detail were chosen using the following method:

1. The inhalation risk was calculated for each chemical

2. The total inhalation risk was calculated for all carcinogens

3. The cumulative percent of the total cancer risk was calculated

4. Compounds which contributed to 99% of the cumulative risk were retained for the finalized screening list

5. Carcinogens which cause public concern were added to this list

The list of carcinogens appearing on the finalized screening list are noted in the section which follows.

Although dioxins are considered to have some potential to be human carcinogens, their biological mode of action suggests that a threshold probably exists for this effect. Evidence from studies of the non-carcinogenic effects of dioxins, particularly reproductive toxicity suggests that this non-carcinogenic effect may be as important as any carcinogenic potential. For these reasons, the toxicity of dioxins was evaluated using the ER approach in conjunction with a very stringent exposure limit.

Figure A-I-1 is a flowchart of the above process.

Non-carcinogens and Dioxins (NC) - The final list of non-carcinogens (and dioxins) to be assessed in detail were chosen using the following method:

1. The exposure ratio was calculated for each NC

2. The total exposure ratio was calculated for all NC

3. Compounds which contributed to 0.99 of the cumulative exposure ratio were retained for the finalized screening list

4. NC which cause public concern were added to this list
The list of NC appearing on the finalized screening list are noted in the section which follows.

Figure A-I-2 is a flowchart of the above process.

Results of the ranking and screening of the MEI health risks of all measurable chemicals in MSWI emissions are shown in Tables A-I-1 and A-I-2.

A-I-1.4.3.5 Finalized Screening List

Of the many chemicals monitored at both facilities, 15 chemicals remained on the final screening list: six are carcinogens, the remaining nine chemicals are NC (Table A-I-3).

Chromium, benzene, cadmium and vinyl chloride contributed to over 99% of the cumulative percent cancer risk. Arsenic and dioxins were added to the list of carcinogens for further assessment due to public concern.

The following compounds contributed to 99% of the cumulative exposure ratio: nitrogen oxides, silicon, hydrogen chloride, mercury, sulphur dioxide, iron and tin. Particulate emissions and lead were added to the list of non-carcinogens for further assessment due to public concern.

Table A-I-4 contains the toxicological information for the carcinogens and non-carcinogens which remain on the post-screening list.

A-I-1.5 Protocol for Assessing Exposure and Health Risks of Critical Chemicals in MSWI Emissions

In the second part of the risk assessment, potential adverse health effects of the critical chemicals screened from the first list of stack emissions were characterized in more detail. Air, soil and local produce exposures were estimated using equations in CAPCOA (1993). Some CAPCOA default values were revised to suit specific chemicals and Ontario procedures for assessing exposure (A-I-1.5.2). Exposures to tapwater and normal food items were not considered. Only inhalation exposures were calculated for particulates, acid gases and airborne materials which present most risk when inhaled (Si, Fe, P, Sn).

A-I-1.5.1 CAPCOA Modelling

The model devised by CAPCOA (1993) was used to assess exposure to the critical chemicals. This model was used to calculate levels of chemicals in air, vegetation and soil. The intake was modelled for the following routes of exposure: inhalation, dermal absorption (due to contact with soil), soil ingestion and food. Table A-I-5 lists the chemical specific constants used in the modelling equations. Parameters which differ from the default values provided by CAPCOA are noted in the relevant sections below.

A-I-1.5.1.1 *Determining Levels in Air* (Maximum GLC - annual average)

Stack emissions to GLC - The Peel dispersion factors (see Table A-II-2) were used to estimate the maximum annual average POI/GLC. These maximum annual average GLC were used as inputs to the CAPCOA model (Table A-I-6).

A-I-1.5.1.2 *Determining Levels in Soil*

Soil levels were calculated using the equations given on pp E-II-2 to E-II-3 (CAPCOA, 1993). The following parameters were changed:

Parameter	Default given in CAPCOA	Value used in this Assessment
Dep-rate (benzene, vinyl chloride)	0.02 (controlled source)	0.0035 (McKone and Daniels, 1991)
Soil Mixing Depth	0.01m (playground), 0.15m (agricultural)	0.05m (McIlveen and McLaughlin, 1993)
Soil Bulk Density	1333 kg/m3	1800 kg/m3 (Copeland et al., 1994)
Total time of exposure	70 years	20 years - see text below

The lifetime of a municipal waste incinerator is usually 20 years. The CAPCOA model was modified in two ways in order to account for the lifetime of the facility. First, the exposure was limited to twenty years (the lifetime of the facility). The second assumption involved the intake of fruits and vegetables and is discussed in the following section. Results are shown in Table A-I-7.

A-I-1.5.1.3 *Determining Levels in Vegetation*

Vegetation levels were calculated using the equations given on pp E-II-8 to E-II-10 of the CAPCOA manual (CAPCOA, 1993). The following parameters were changed:

Parameter	Default given in CAPCOA	Value used in this Assessment
IF Leafy crops Vine crops	 0.2 0.1	 0.39 0.073 (Copeland etal., 1994)
Y	2	4.4
T	45-90	90
Ctrans (all chemicals)	Cs*UF2	assumed Ctrans=0 (for benzene and vinyl chloride).*

* For benzene and vinyl chloride the parameter "k" was replaced with R_v (MacKay et al., 1992). The equation to calculate R_v is

$$R_v = \frac{V_{da}}{Y} \times \frac{H}{RT} \times \frac{10^3}{0.9 + 0.1K_{ow}}$$

where R_v is in units of $(\text{day})^{-1}$ and the parameters are as follows:

V_{da}=302 m/day
Y=4.4 kg/m2
H=4155 torr*m^3/mol for benzene and 446430 for vinyl chloride
R=62.4 (torr*L)/(mol*K)
T=298 K
K_{ow} listed in Table A-I-3

When considering intake of vegetation, transfer to leaves and fruits (or vegetables) from root uptake was considered for a further 10 years. However, because the facility was closed during this period of time, it was assumed that there would be a loss of deposited chemicals. The loss was assumed to obey first-order kinetics (Borgert *et al*, 1995) according to the following model: $C=C_oe^{-kt}$, where C represents the instaneous concentration, C_o is the concentration in soil after 20 years of deposition from the incinerator and k is the loss constant and t is time. A constant concentration was calculated by finding the mean value of the above loss equation (Grossman, 1981). Results are shown in Table A-I-8.

A-I-1.5.1.4 *Receptor Characterization*

The receptor characteristics are summarized in Table A-I-9. A female receptor was used because at all ages, females have a higher intake of a given toxicant. Thus, the use of a female receptor leads to a more conservative estimate of exposure than the use of a male receptor. The receptor characteristics which have a distribution are shown in shaded cells of Table A-I-9. The distributions associated with these parameters are included in Appendix B-III.

Also, it was assumed that the receptor was born near the incinerator when it first opened and that she remained there during the entire period of operation (20 years) and 10 years after the incinerator closed.

A-I-1.5.1.5 *Estimation of Exposure through Inhalation*

Intake of chemicals by inhalation was determined using the following equation:

$$\frac{(infiltrationrate*GLC*RR*frac_{indoors})+(RR*GLC*frac_{outdoors})}{ABW*1000}$$

where the infiltration rate=0.75, the fraction of time spent indoors=0.8 and the fraction of time spent outdoors=0.2 (Hawley, 1985, Hawkins, 1991). Since a composite receptor was used, the appropriate respiration rate (RR) and average body weight (ABW) was used for each stage of life and the lifetime average daily dose (LADD) was calculated for each chemical. Results are shown in Table A-I-10.

A-I-1.5.1.6 *Estimation of Exposure through Dermal Absorption*

Intake through dermal absorption was calculated using the equation given on p. E-II-16 (CAPCOA, 1993). The numerator was multiplied by a factor of 0.67 to account for the fact that during winter months skin does not come in direct contact with soil due to snow cover, winter clothing, etc. Since a composite receptor was used, the appropriate surface area (SA) and average body weight (ABW) was used for each stage of life and the lifetime average daily dose (LADD) was calculated for each chemical. Results are shown in Table A-I-11.

A-I-1.5.1.7 *Estimation of Exposure through Ingestion of Soil*

Intake through ingestion of soil was calculated using the equation given on pp E-II-16 (CAPCOA, 1993). The bioavailability (BIO) was only provided for dioxins in Table 1 of the CAPCOA document (CAPCOA, 1993); thus BIO=1 was assumed for all other compounds. Since a composite receptor was used, the appropriate soil ingestion rate (Is) and average body weight (ABW) was used for each stage of life and the lifetime average daily dose (LADD) was calculated for each chemical. Results are shown in Table A-I-12.

A-I-1.5.1.*8* *Estimation of Exposure through Ingestion of Food*

Intake through ingestion of soil was calculated using the equation given on pp E-II-18 (CAPCOA, 1993). The denominator for this equation should have been written as (ABW*1000) - note that the brackets are missing in the CAPCOA document. The fraction of plant type F which is homegrown (L) is assumed to be 0.52 (U.S. EPA, 1990). Since a composite receptor was used, the appropriate ingestion of plant type F (If) and average body weight (ABW) was used for each stage of life and the lifetime average daily dose (LADD) was calculated for each chemical.

As explained above, the intake of a compound from produce for females greater than 20 years, was the mean concentration from 20-30 years and the period of intake was 10 years after the closing of the incinerator. Results are shown in Table A-I-13.

A-I-1.5.1.*9* *Calculation of the Lifetime Average Daily Dose (LADD)*

The lifetime average daily dose is a weighted mean. The weighting factors are derived from the proportion of a total lifetime for a particular stage. The lifetime average daily dose (LADD) for carcinogens was calculated as follows:

$$\frac{(Stage_1 \times 182.5) + (Stage_2 \times 1277.5) + (Stage_3 \times 2920) + (Stage_4 \times 2920) + (Stage_5 \times 3650)}{25550}$$

where Stage 1 represents 0 to 6 months
Stage 2 represents 7 months to 4 years
Stage 3 represents 5 to 11 years
Stage 4 represents 12-19 years
and Stage 5 represents 20-30 years (considering a person who resides at the site 10 years after the facility closes).
Note: 25550 is the number of days in 70 years (ie. length of the average lifetime)

For non-carcinogenic toxicants, the denominator used was 10950 days (= 30 years). Results are shown in Table A-I-14.

A-I-1.5.3 Probabilistic Modelling of Individual Risk

The probabilistic analysis was conducted only for the exposure component of this assessment while point values were used for the toxicological component; the toxicological point values were health-based guidelines (Section A-I-1.4.3.1). Probabilistic exposure estimates and potential health risks were computed with commercial software (Crystal Ball Version 3.0, Decisioneering Inc., Denver, CO) running on Excel (Microsoft Excel Version 5.0). A total of 10,000 iterations were conducted using a Monte-Carlo sampling procedure in order to solve forecast cells using distributional assumptions for various parameters. Parameters known to be correlated (e.g. body weight and inhalation rate) were arbitrarily allocated a correlation coefficient of 0.67.

Tables 5,6 and 7 of the main report show results of the probabilistic modelling for all compounds and compares the probabalistic estimates to the deterministic risk estimates. The assumptions used for various parameters are provided in Appendix III.

A-I-1.5.4 Modelling of Population Risk

Calculating the number of expected cases of cancer due to emissions from an incinerator is described in appendix II.
The results of the population-based risk estimates are presented in Table 7 of the main report.

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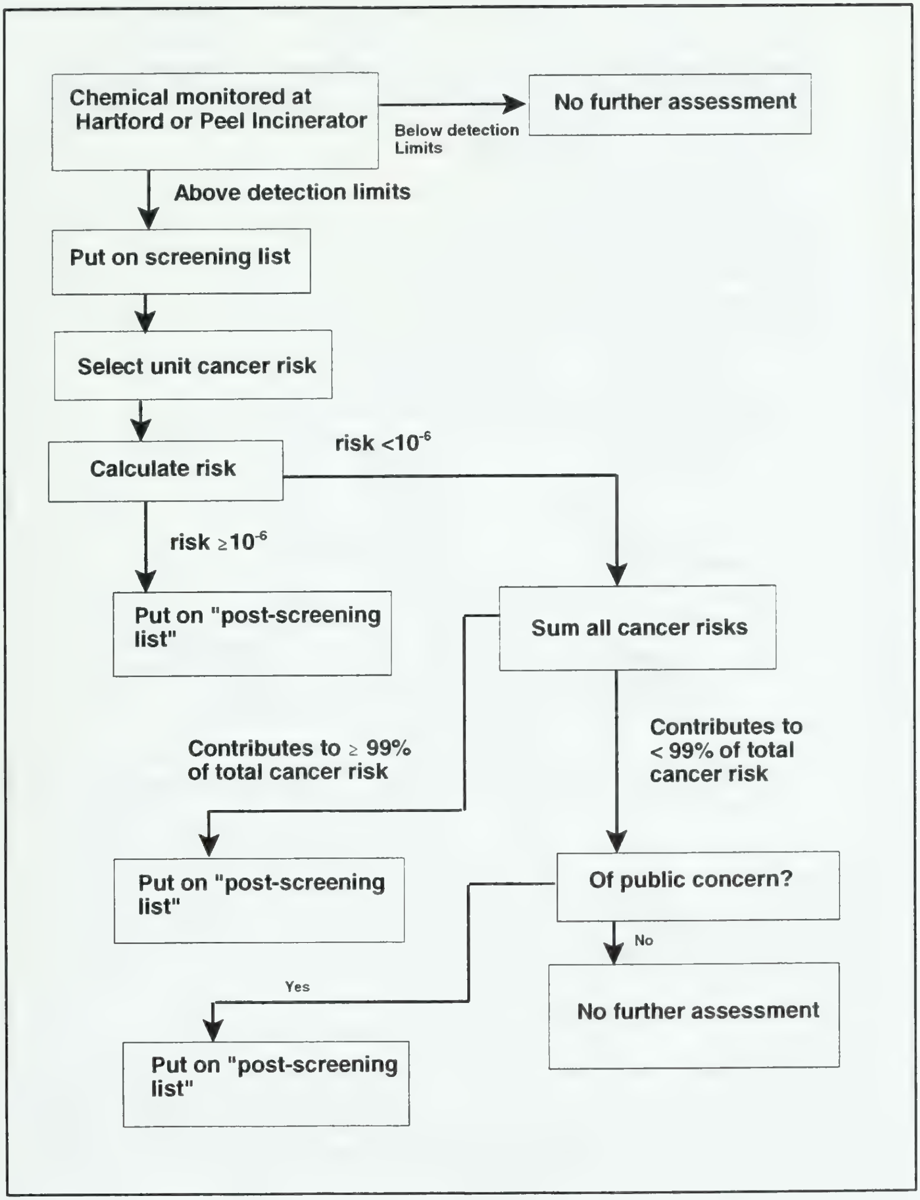
Chemical monitored at Hartford or Peel Incinerator
Below detection Limits
No further assessment
Above detection limits
Put on screening list
Select unit cancer risk
Calculate risk
risk <10⁻⁶
risk ≥10⁻⁶
Put on "post-screening list"
Sum all cancer risks
Contributes to ≥ 99% of total cancer risk
Contributes to < 99% of total cancer risk
Put on "post-screening list"
Of public concern?
No
Yes
No further assessment
Put on "post-screening list"

Figure A-I-1. Flowchart of Screening Process for Carcinogens in MWSI emissions.

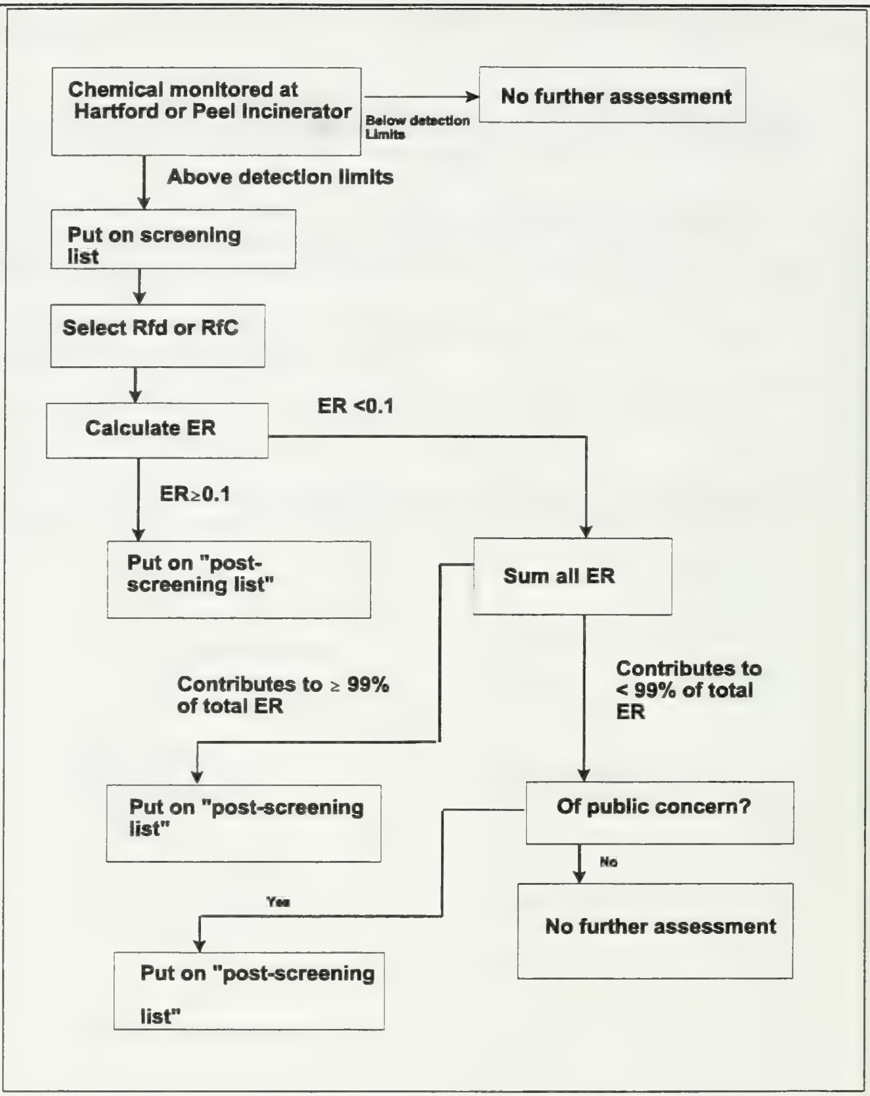
Chemical monitored at Hartford or Peel Incinerator
Below detection Limits
No further assessment
Above detection limits
Put on screening list
Select Rfd or RfC
Calculate ER
ER <0.1
ER≥0.1
Put on "post-screening list"
Sum all ER
Contributes to ≥ 99% of total ER
Contributes to < 99% of total ER
Put on "post-screening list"
Of public concern?
No
Yes
No further assessment
Put on "post-screening list"

Figure A-I-2. Flowchart of Screening Process for Non-Carcinogens in MWSI Emissions

Figure A-I-2. Flowchart of Screening Process for Non-carcinogens in MSWI Emissions.

Table A-I-1. Ranking of Exposure Ratios of Measured Systemic Toxicants in Emissions from a PEEL-like Incinerator to Screen for Critical Chemicals.

Parameter	Emission rate (ug/s)	GLCmax24h (ug/m3)	RfC (ug/cu.m)	Intake Factor (ug/kg/d)	Critical Intake(ug/kg/d)	Calculated Exposure Ratio	Cumulative Exposure Ratio	Endpoint	Reference
Silicon	1187257	2.02E+00	5		0.6632	0.4037	0.4031	pulmonary function - silicosis	NATICH, 1992
Nitrogen Oxides	19044242	3.24E+01	100		10.6376	0.3238	0.7264	pulmonary function	HEAST, 1992
Hydrogen Chloride	790781	1.34E+00	7		0.4417	0.1920	0.9181	hyperplasia of upper resp. tract	IRIS, 1994
Mercury	12632	2.15E-02		0.3	0.0071	0.0235	0.9416	neurotoxicity	HEAST, 1992
Sulphur Dioxide	1051144	1.79E+00	80		0.5871	0.0223	0.9639	pulmonary function	HEAST, 1992
Iron	6441	1.09E-02	1		0.0036	0.0109	0.9749	pulmonary function - siderosis	NATICH, 1992
Tin	484	8.23E-04	0.1		0.0003	0.0082	0.9831	pulmonary function - stannosis	NATICH, 1992
Particulate emissions	117740	2.00E-01	50		0.0658	0.0040	0.9871	pulmonary function	HEAST, 1992
Phosphorus	1266	2.15E-03	0.63	0.207	0.0007	0.0034	0.9905	mucous membrane irritant, bone disorders	NATICH, 1992
Manganese	294	5.00E-04		0.05	0.0002	0.0033	0.9938	(inhal) - impairment of neuro-behavioral function	IRIS, 1994
Cobalt	159	2.70E-04	0.1		0.0001	0.0027	0.9965	not available	MOEE, 1994
Vanadium	32	5.44E-05	0.03		0.0000	0.0018	0.9983	calculated from 24 hr. ave	NATICH, 1992
Magnesium	1000	1.70E-03	2.74		0.0006	0.0006	0.9989	calculated from 24 hr ave	NATICH, 1992
Aluminum	1093	1.86E-03	5		0.0006	0.0004	0.9993	not available	NATICH, 1992
Lead	679	1.15E-03		1.85	0.0004	0.0002	0.9995	CNS effects in children	OMOEE, 1994
Copper	241	4.10E-04	2.4		0.0001	0.0002	0.9996	not available	CAPCOA, 1993
Bromomethane	323	5.49E-04	5		0.0002	0.0001	0.9998	nasal cavity lesions, epithelial hyperplasia of forestomach	IRIS, 1994
Nickel	2396	4.07E-03		20	0.0013	0.0001	0.9998	dec. body and organ weights	IRIS, 1994
Antimony	42.2	7.17E-05		0.4	0.0000	0.0001	0.9999	longevity, blood glucose and cholesterol	IRIS, 1994
Total Xylenes	5538	9.41E-03		144	0.0031	0.0000	0.9999	maternal and fetal toxicity	CEPA, 1993
Boron	2945	5.01E-03		90	0.0016	0.0000	0.9999	testicular atrophy, spermatogenic arrest	IRIS, 1994
Molybdenum	117	1.99E-04		5	0.0001	0.0000	1.0000	inc. uric acid levels	IRIS, 1994
Silver	92	1.56E-04		5	0.0001	0.0000	1.0000	argyria	IRIS, 1994
Total Tetrachlorobenzenes	5.6	9.52E-06		0.21	0.0000	0.0000	0.9999	thyroid follicular hypertrophy	CEPA, 1993

Table A-I-1. Ranking of Exposure Ratios of Measured Systemic Toxicants in Emissions from a PEEL-like Incinerator to Screen for Critical Chemicals (continued).

Parameter	Emission rate (ug/s)	GLCmax24h (ug/m3)	Rf C (ug/cu.m)	Intake Factor (ug/kg/d)	Critical Intake(ug/kg/d)	Calculated Exposure Ratio	Cumulative Exposure Ratio	Endpoint	Reference
Zinc	5487	9.33E-03		300	0.0031	0.0000	1.0000	dec. SOD in erythrocytes in adult female	IRIS, 1994
Selenium	52.3	8.89E-05		5	0.0000	0.0000	1.0000	clinical seleniosis	IRIS, 1994
Total Dichlorophenols	7.3	1.24E-05	3	3	0.0000	0.0000	1.0000	calculated from 30 min. ave.	NATICH, 1992
Pentachlorobenzene	3.7	6.29E-06		0.5	0.0000	0.0000	1.0000	hepatocyte abnormalities	CEPA, 1993
Total Trichlorobenzenes	5.4	9.18E-06		0.77	0.0000	0.0000	1.0000	inc. liver wt/body wt ratio, histopathological chngs to liv., thy., kid.	CEPA, 1993
Trichlorofluoromethane	1579	2.68E-03		300	0.0009	0.0000	1.0000	survival, histopathology	IRIS, 1994
Dichlorodifluoromethane	711	1.21E-03	600		0.0004	0.0000	1.0000	dec. body weight	IRIS, 1994
Toluene	464	7.89E-04	400		0.0003	0.0000	1.0000	CNS	IRIS, 1994
Biphenyl	82	1.39E-04		50	0.0000	0.0000	1.0000	kidney damage	IRIS, 1994
o-Terphenyl	2.3	3.91E-06	5		0.0000	0.0000	1.0000	not available	NATICH, 1992
Barium	58	9.86E-05		70	0.0000	0.0000	1.0000	inc. blood pressure	IRIS, 1994
Fluoride	21.2	3.60E-05		60	0.0000	0.0000	1.0000	cosmetic: objectionable dental fluorosis	IRIS, 1994
Pyrene	1	1.70E-06		3	0.0000	0.0000	1.0000	kidney effects	IRIS, 1994
1,4-Dichlorobenzene	23.7	4.03E-05	240		0.0000	0.0000	1.0000	liver, kidney, proteinuria	CEPA, 1993
Total tetrachlorophenols	4.8	8.16E-06		30	0.0000	0.0000	1.0000	inc. liver wt and centrilobular hypertrophy	IRIS, 1994
I,I,I-Trichloroethane	13.1	2.23E-05	270		0.0000	0.0000	1.0000	liver	HEAST, 1992
Strontium	31	5.27E-05		600	0.0000	0.0000	1.0000	rachitic bone	IRIS, 1994
Fluorene	1.3	2.21E-06		40	0.0000	0.0000	1.0000	dec. RBC, packed cell volume and hemoglobin	IRIS, 1994
Ethylbenzene	9.5	1.62E-05	1000	100	0.0000	0.0000	1.0000	liver and kidney toxicity	IRIS, 1994
Fluoranthene	1.1	1.87E-06		40	0.0000	0.0000	1.0000	nephropathy, inc. liver weight, hematological alterations and clinical effects	IRIS, 1994
1,2-Dichlorobenzene	2	3.40E-06		430	0.0000	0.0000	1.0000	tubular regeneration of kidney	CEPA, 1993
1,3-Dichlorobenzene	2	3.40E-06	1310		0.0000	0.0000	1.0000		NYDOH, 1992
Tetrachloroethene	4.4	7.48E-06	100000		0.0000	0.0000	1.0000	multi-site	CEPA, 1993

Stack Parameters based on maxima from 1993 & 1994 testing of Peel + positives from Hartford (1 boiler) tests for some Peel ND

All emission rate values corrected for stack temp. (160 Celsius) to ground level (25 Celsius)

GLCmax **24hr values** calculated using Peel stack data (Table A-II-1) and **worst case** Ontario met. data (Buttonville (Table A-II-2))

Table A-I-2. Ranking of Risks of Measured Carcinogens in Emissions from a PEEL-like Incinerator to Screen for Critical Chemicals..

Parameter	Emission rate (ug/s)	GLCmax24h (ug/m3)	Risk Factor	Critical Intake (ug/kg/d)	Calculated Inhalation Risk or Risk Index	Cumulative Percent Risk8	Endpoint	Reference
Chromium	149	2.53E-04	4.1E-2/ug/kg/d	8.32E-05	3.41E-06	82.70	lung cancer	HEAST, 1992
Benzene	24890	4.23E-02	8.3E-6/ ug/cu. m	1.39E-02	3.51E-07	91.21	leukemia	IRIS, 1994
Cadmium	71.2	1.21E-04	1.8E-3/ ug/cu. m	3.98E-05	2.18E-07	96.49	upper resp. tract	IRIS, 1994
Vinyl Chloride	400	6.80E-04	8.4E-5/ug/cu. m	2.23E-04	5.71E-08	97.88	liver tumors	HEAST, 1992
Arsenic	21.8	3.71E-05	3.53E-3/ug/kg/d	1.22E-05	4.30E-08	98.92	internal cancers (kidney, lung, liver, bladder)	OMOEE, 1994
Beryllium	9.2	1.56E-05	2.4E-3/ug/cu. m	5.14E-06	3.75E-08	99.83	gross tumors (all sites)	IRIS, 1994
Chloromethane	1441	2.45E-03	1.8E-6/ug/cu.m	8.05E-04	4.41E-09	99.94	kidney tumors	HEAST, 1992
Dibromoethane	3.9	6.63E-06	2.2E-4/ ug/cu. m	2.18E-06	1.46E-09	99.97	squamous cell carcinoma of forestomach	IRIS, 1994
1,1-Dichloroethene	5.2	8.84E-06	5E-5/ug/cu. m	2.90E-06	4.42E-10	99.98	kidney adeno-carcinomas	IRIS, 1994
Hexachlorobenzene	0.4	6.80E-07	4.6E-4/ug/cu. m	2.23E-07	3.13E-10	99.99	hepatocellular carcinoma	IRIS, 1994
Pentachlorophenol	2.8	4.76E-06	1.2E-4/ug/kg/d	1.56E-06	1.88E-10	100.00	hepatocellular carcinoma	IRIS, 1994
Total Trichlorophenols	13.4	2.28E-05	3.1E-6 /ug/cu. m	7.48E-06	7.06E-11	100.00	leukemia (male rats)	IRIS, 1994
Benzo(b)fluoranthene	0.15	2.55E-07	8.03E-4/ug/kg/d	8.38E-08	6.73E-11	100.00	liver tumors	MOEE, 1996
Trichloroethene	13.5	2.30E-05	2E-6/ ug/cu. m	7.54E-06	4.59E-11	100.00	not available	CAPCOA, 1993
Phenanthrene	6.2	1.05E-05	4.67E-6/ug/kg/d	3.46E-06	1.62E-11	100.00	liver tumors	MOEE, 1996
Total Chlorobiphenyl	1.9	3.23E-06	5.3E-6/ug/kg/d	1.06E-06	5.62E-12	100.00	liver tumors	MOL, 1984
Total Dioxins (I-TEQ)	0.00814	1.38E-08	10 pg/kg/d	4.55E-09	4.55E-04		hepatocellular carcinoma	CEPA, 1990

Stack Parameters based on maxima from 1993 & 1994 testing of Peel + positives from Hartford (1 boiler) tests for some Peel ND

All emission rate values corrected for stack temp. (160 Celsius) to ground level (25 Celsius)

GLCmax **24hr** values calculated using Peel stack data (Table A-II-1) and **worst case** Ontario met. data (Buttonville)(Table A-II-2).

Table A-I-3: Chemicals Assessed in Detail for Possible Adverse Health Effects

Carcinogens	**Non-carcinogens**
Chromium	Nitrogen Oxides
Benzene	Silicon
Cadmium	Hydrogen Chloride
Vinyl Chloride	Mercury
Arsenic	Sulphur Dioxide
Dioxins	Iron
	Tin
	Particulate emissions
	Lead

TableA-I-4. Toxicity Factors for Screened Chemicals

Critical Chemicals					
	RfC (ug/cu.m)	Risk Factor	Intake Factor (ug/kg/d)	Endpoint	Reference
Chromium		4.1 E-2 /ug/kg/d		carcinogenic	HEAST, 1992
Chromium	0.000083			not available	NATICH, 1992
Benzene		2.9 E-5 /ug/kg/d		leukemia	IRIS, 1994
Benzene		8.3E-6 /ug/cu. m		leukemia	IRIS, 1994
Cadmium			0.5	renal	IRIS, 1994
Cadmium			1	renal	IRIS, 1994
Cadmium		1.8E-3 /ug/cu. m		carcinogenic	IRIS, 1994
Vinyl Chloride		5.26E-4 /ug/kg/d		lung and liver tumors	HEAST, 1992
Vinyl Chloride		8.4E-5 /ug/cu. m		liver tumors	HEAST, 1992
Arsenic		3.53E-3 /ug/kg/d		carcinogenic	OMOEE, 1994
Total I-TEQ			3.30e-07	liver tumors	CEPA, 1990
Nitrogen Oxides	100			pulmonary	HEAST, 1992
Silicon	5			pneumoconiotic	NATICH, 1992
Hydrogen Chloride	7			pulmonary	IRIS, 1994
Mercury	3.00e-01			neurotoxic	HEAST, 1992
Sulphur Dioxide	80			not available	HEAST, 1992
Iron	1			pneumoconiotic	NATICH, 1992
Tin	0.1			pneumoconiotic	NATICH, 1992
Particulate emissions	50			not available	HEAST, 1992
Lead			1.85	neurotoxic	OMOEE, 1994

Table A-I-5. Constants used for CAPCOA Models.

Chemical	Soil half-life (days)	Uptake Factors			Uptake Para.			Derm. Abs. Factor	GI Abs. Factor	Bioavail.
		Leaf	Vine	Root	Kow	Koc	Foc			
Chromium	1.00E+08	8.00E-04	6.00E-04	1.00E-03				1.00E-02	1.00E+00	1.00E+00
Benzene	1.60E+01	2.57E-01	2.57E-01	2.57E-01	1.35E+02	8.30E+01	1.00E-01	3.00E-02	1.00E+00	1.00E+00
Cadmium	1.00E+08	6.00E-02	2.00E-02	4.00E-02				2.00E-03	1.00E+00	1.00E+00
Vinyl Chloride	1.80E+02	3.28E-01	3.28E-01	3.28E-01	1.40E+01	3.20E+01	1.00E-01	3.00E-02	1.00E+00	1.00E+00
Arsenic	1.00E+08	4.00E-03	9.00E-04	2.00E-03				1.00E-03	1.00E+00	1.00E+00
Dioxins*	3.29E+03	1.52E-02	1.52E-02	1.52E-02	4.27E+06	2.51E+06	1.00E-01	2.00E-02	1.00E+00	4.30E-01
Nitrogen Oxides									1.00E+00	1.00E+00
Silicon									1.00E+00	1.00E+00
Hydrogen Chloride									1.00E+00	1.00E+00
Mercury	4.90E+03	9.00E-02	3.00E-02	2.00E-02				1.00E-02	1.00E+00	1.00E+00
Sulphur Dioxide									1.00E+00	1.00E+00
Iron	1.00E+08	6.00E-02	2.00E-02	4.00E-02				2.00E-03	1.00E+00	1.00E+00
Tin	1.00E+08	6.00E-02	2.00E-02	4.00E-02				2.00E-03	1.00E+00	1.00E+00
Particulate Emissions									1.00E+00	1.00E+00
Lead	1.00E+08	5.00E-03	1.00E-03	2.00E-03				1.00E-03	1.00E+00	1.00E+00

Table A-I-6. Results of Modified CAPCOA Exposure Assessment Model - Estimation of Air Concentrations.

Chemical	GLCmaxAA (ug/m3)	Deposition (ug/m2/d)
Chromium	1.12E-05	6.11E-03
Benzene	1.87E-03	1.02E+00
Cadmium	5.34E-06	2.92E-03
Vinyl Chloride	3.00E-05	1.64E-02
Arsenic	1.64E-06	8.94E-04
Dioxins	6.11E-10	3.34E-07
Nitrogen Oxides	1.43E+00	7.82E+02
Silicon	8.90E-02	4.87E+01
Hydrogen Chloride	5.93E-02	3.24E+01
Mercury	9.47E-04	5.18E-01
Sulphur Dioxide	7.88E-02	4.31E+01
Iron	4.83E-04	2.64E-01
Tin	3.63E-05	1.99E-02
Particulate emissions	8.83E-03	4.83E+00
Lead	5.09E-05	2.79E-02

GLCmaxAA (annual average) values calculated using Peel stack data (Table A-II-1) and worst case Ontario met. data (Buttonville) (Table A-II-2).

Table A-I-7. Results of Modified CAPCOA Exposure Assessment Model - Estimation of Soil Concentrations

ug/kg

Chemical	X	Soil Conc. (20 years)	Time decay (days-1)	Soil Conc. (ave: 20-30 years)
Chromium	1.85E-01	2.48E-01	6.93E-09	2.48E-01
Benzene	7.28E+03	2.61E-01	4.33E-02	7.46E-141
Cadmium	1.85E-01	1.18E-01	6.93E-09	1.18E-01
Vinyl Chloride	7.04E+03	4.57E-02	3.85E-03	2.01E-15
Arsenic	1.85E-01	3.63E-02	6.93E-09	3.63E-02
Dioxins	3.58E+03	8.61E-06	2.11E-04	1.29E-06
Nitrogen Oxides				
Silicon				
Hydrogen Chloride				
Mercury	2.75E+03	1.53E+01	1.41E-04	4.26E+00
Sulphur Dioxide				
Iron	1.85E-01	1.07E+01	6.93E-09	1.07E+01
Tin	1.85E-01	8.05E-01	6.93E-09	8.05E-01
Particulate emissions				
Lead	1.85E-01	1.13E+00	6.93E-09	1.13E+00

Table A-I-8. Results of Modified CAPCOA Exposure Assessment Model - Estimation of Concentrations in Vegetation

				First 20 years				
Chemical	Cdepv Leaf	Cdepv Vine	Ctrans Leaf	Ctrans Vine	Ctrans Root	Cf Leaf	Cf Vine	Cf Root
Chromium	7.80E-03	1.46E-03	1.98E-04	1.49E-04	2.48E-04	8.00E-03	1.61E-03	2.48E-04
Benzene	8.44E-05	1.58E-05				8.44E-05	1.58E-05	
Cadmium	3.73E-03	6.98E-04	7.11E-03	2.37E-03	4.74E-03	1.08E-02	3.07E-03	4.74E-03
Vinyl Chloride	2.12E-08	3.98E-09				2.12E-08	3.98E-09	
Arsenic	1.14E-03	2.14E-04	1.45E-04	3.26E-05	7.25E-05	1.29E-03	2.46E-04	7.25E-05
Dioxins	4.26E-07	7.98E-08	1.31E-07	1.31E-07	1.31E-07	3.15E-07	1.66E-07	1.31E-07
Nitrogen Oxides								
Silicon								
Hydrogen Chloride								
Mercury	6.61E-01	1.24E-01	1.38E+00	4.60E-01	3.06E-01	2.04E+00	5.83E-01	3.06E-01
Sulphur Dioxide								
Iron	3.37E-01	6.31E-02	6.43E-01	2.14E-01	4.29E-01	9.80E-01	2.77E-01	4.29E-01
Tin	2.53E-02	4.74E-03	4.83E-02	1.61E-02	3.22E-02	7.37E-02	2.08E-02	3.22E-02
Particulate emissions								
Lead	3.56E-02	6.65E-03	5.65E-03	1.13E-03	2.26E-03	4.12E-02	7.78E-03	2.26E-03

Table A-I-8.
Results of Modified CAPCOA Exposure Assessment Model - Estimation of Concentrations in Vegetation.

	Last 10 years							
Chemical	Cdepv Leaf	Cdepv Vine	Ctrans Leaf	Ctrans Vine	Ctrans Root	Cf Leaf	Cf Vine	Cf Root
Chromium	0	0	1.98E-04	1.49E-04	2.48E-04	1.98E-04	1.49E-04	2.48E-04
Benzene	0	0	0	0	0			
Cadmium	0	0	7.11E-03	2.37E-03	4.74E-03	7.11E-03	2.37E-03	4.74E-03
Vinyl Chloride	0	0	0	0	0			
Arsenic	0	0	1.45E-04	3.26E-05	7.25E-05	1.45E-04	3.26E-05	7.25E-05
Dioxins	0	0	1.96E-08	1.96E-08	1.96E-08	1.96E-08	1.96E-08	1.96E-08
Nitrogen Oxides								
Silicon								
Hydrogen Chloride								
Mercury	0	0	3.83E-01	1.28E-01	8.52E-02	3.83E-01	1.28E-01	8.52E-02
Sulphur Dioxide								
Iron	0	0	6.43E-01	2.14E-01	4.29E-01	6.43E-01	2.14E-01	4.29E-01
Tin	0	0	4.83E-02	1.61E-02	3.22E-02	4.83E-02	1.61E-02	3.22E-02
Particulate emissions								
Lead	0	0	5.65E-03	1.13E-03	2.26E-03	5.65E-03	1.13E-03	2.26E-03

Table A-I-9: Receptor Characteristics

Age	Body Weight (kg)	Air intake (m^3/d)	Soil Intake (mg/d)	Active Surface Area (cm^2)	Leafy Intake (kg/d)	Vine Intake (kg/d)	Root Intake (kg/d)	Fruits (kg/d)	Vegetable Intake (kg/d)
0-6 m	6.28	1.21	35.0	1750	0.0154	0.150	0	0.110	0.040
7 m-4y	13.0	11.0	91.0	2140	0.0372	0.311	0.061	0.262	0.049
5y-12y	26.8	15.9	91.0	2380	0.0475	0.376	0.099	0.305	0.071
13y-20y	50.0	26.1	250.0	3630	0.0456	0.382	0.074	0.287	0.095
21+ y	61.5	21.4	100.0	3400	0.0428	0.350	0.076	0.236	0.114

Reference: IWA, 1994

Table A-I-10.
Results of Modified CAPCOA Exposure Assessment Model - Estimation of Exposure through Inhalation.

Chemical	0-6 m (ug/kg/d)	7 m-4y (ug/kg/d)	5-12 (ug/kg/d)	13-20 (ug/kg/d)	21+ (ug/kg/d)	**LADD** (ug/kg/d)
Chromium	2.15E-06	9.46E-06	6.63E-06	5.83E-06	3.89E-06	1.91E-06
Benzene	3.60E-04	1.58E-03	1.11E-03	9.74E-04	6.50E-04	3.19E-04
Cadmium	1.03E-06	4.52E-06	3.17E-06	2.79E-06	1.86E-06	9.14E-07
Vinyl Chloride	5.78E-06	2.54E-05	1.78E-05	1.57E-05	1.04E-05	5.13E-06
Arsenic	3.15E-07	1.38E-06	9.70E-07	8.53E-07	5.69E-07	2.80E-07
Dioxins	1.18E-10	5.17E-10	3.62E-10	3.19E-10	2.12E-10	1.04E-10
Nitrogen Oxides	2.76E-01	1.21E+00	8.48E-01	7.46E-01	4.98E-01	8.57E-01
Silicon	1.72E-02	7.53E-02	5.28E-02	4.65E-02	3.10E-02	5.33E-02
Hydrogen Chloride	1.14E-02	5.02E-02	3.52E-02	3.10E-02	2.06E-02	3.55E-02
Mercury	1.83E-04	8.02E-04	5.62E-04	4.95E-04	3.30E-04	5.67E-04
Sulphur Dioxide	1.52E-02	6.67E-02	4.68E-02	4.12E-02	2.74E-02	4.72E-02
Iron	9.31E-05	4.09E-04	2.87E-04	2.52E-04	1.68E-04	2.89E-04
Tin	6.99E-06	3.07E-05	2.15E-05	1.89E-05	1.26E-05	2.17E-05
Particulate emissions	1.70E-03	7.47E-03	5.24E-03	4.61E-03	3.07E-03	5.29E-03
Lead	9.81E-06	4.31E-05	3.02E-05	2.66E-05	1.77E-05	3.05E-05

Table A-I-11.
Results of Modified CAPCOA Exposure Assessment Model - Estimation of Exposure through Dermal Absorption

Chemical	0-6 m (ug/kg/d)	7 m-4y (ug/kg/d)	5-12 (ug/kg/d)	13-20 (ug/kg/d)	21+ (ug/kg/d)	**LADD (ug/kg/d)**
Chromium	1.19E-07	1.41E-07	7.63E-08	6.24E-08	4.75E-08	1.07E-07
Benzene	3.78E-07	4.47E-07	2.41E-07	1.97E-07	4.29E-147	7.51E-08
Cadmium	1.14E-08	1.35E-08	7.29E-09	5.96E-09	4.54E-09	1.02E-08
Vinyl Chloride	6.60E-08	7.82E-08	4.22E-08	3.45E-08	1.16E-21	1.31E-08
Arsenic	1.75E-09	2.07E-09	1.12E-09	9.13E-10	6.95E-10	4.47E-10
Dioxins	8.31E-12	9.83E-12	5.30E-12	4.34E-12	4.93E-13	1.72E-12
Nitrogen Oxides						
Silicon						
Hydrogen Chloride						
Mercury	7.39E-06	8.74E-06	4.72E-06	3.86E-06	8.17E-07	5.55E-06
Sulphur Dioxide						
Iron	1.03E-06	1.22E-06	6.60E-07	5.39E-07	4.11E-07	9.25E-07
Tin	7.76E-08	9.19E-08	4.96E-08	4.05E-08	3.09E-08	6.95E-08
Particulate emissions						
Lead	5.45E-08	6.45E-08	3.48E-08	2.84E-08	2.16E-08	4.87E-08

Table A-I-12.
Results of Modified CAPCOA Exposure Assessment Model - Estimation of Exposure hrough Soil Ingestion.

Chemical	0-6 m ug/kg/day	7 m-4y ug/kg/day	5-12 ug/kg/day	13-20 ug/kg/day	21+ ug/kg/day	**LADD ug/kg/day**
Chromium	6.91E-07	1.74E-06	8.42E-07	1.24E-06	4.03E-07	9.08E-07
Benzene	7.28E-07	1.83E-06	8.86E-07	1.31E-06	1.21E-146	2.33E-07
Cadmium	3.30E-07	8.29E-07	4.02E-07	5.92E-07	1.93E-07	4.34E-07
Vinyl Chloride	1.27E-07	3.20E-07	1.55E-07	2.28E-07	3.27E-21	4.07E-08
Arsenic	1.01E-07	2.54E-07	1.23E-07	1.81E-07	5.90E-08	3.79E-08
Dioxins	1.03E-11	2.59E-11	1.26E-11	1.85E-11	9.00E-13	3.39E-12
Nitrogen Oxides						
Silicon						
Hydrogen Chloride						
Mercury	4.27E-05	1.07E-04	5.20E-05	7.66E-05	6.93E-06	5.01E-05
Sulphur Dioxide						
Iron	2.99E-05	7.50E-05	3.64E-05	5.36E-05	1.74E-05	1.12E-05
Tin	2.24E-06	5.64E-06	2.73E-06	4.03E-06	1.31E-06	8.42E-07
Particulate emissions						
Lead	3.15E-06	7.91E-06	3.84E-06	5.65E-06	1.84E-06	4.14E-06

Table A-I-13. Results of Modified CAPCOA Exposure Assessment Model - Estimation of Exposure through Ingestion of Backyard Produce.

Chemical	Leafy 0-6 m	Vine 0-6 m	Root 0-6 m	Total 0-6 m	Leafy 7 m-4y	Vine 7 m-4y	Root 7 m-4y	Total 7 m-4y	Leafy 5-12	Vine 5-12	Root 5-12	Total 5-12
Chromium	0	0	0	0	5.38E-06	9.05E-06	2.73E-07	1.47E-05	3.33E-06	5.31E-06	2.15E-07	8.85E-06
Benzene	0	0	0	0	5.67E-08	8.88E-08		1.46E-07	3.51E-08	5.21E-08		8.72E-08
Cadmium	0	0	0	0	7.29E-06	1.72E-05	5.22E-06	2.98E-05	4.51E-06	1.01E-05	4.11E-06	1.87E-05
Vinyl Chloride	0	0	0	0	1.43E-11	2.24E-11		3.66E-11	8.85E-12	1.31E-11		2.20E-11
Arsenic	0	0	0	0	8.65E-07	1.38E-06	8.00E-08	2.33E-06	5.36E-07	8.12E-07	6.30E-08	1.41E-06
Dioxins	0	0	0	0	2.12E-10	9.31E-10	1.45E-10	1.29E-09	1.31E-10	5.46E-10	1.14E-10	7.91E-10
Nitrogen Oxides												
Silicon												
Hydrogen Chloride												
Mercury	0	0	0	0	1.37E-03	3.28E-03	3.38E-04	4.99E-03	8.50E-04	1.92E-03	2.66E-04	3.04E-03
Sulphur Dioxide												
Iron	0	0	0	0	6.59E-04	1.56E-03	4.73E-04	2.69E-03	4.08E-04	9.15E-04	3.72E-04	1.69E-03
Tin	0	0	0	0	4.95E-05	1.17E-04	3.55E-05	2.02E-04	3.07E-05	6.87E-05	2.80E-05	1.27E-04
Particulate emissions												
Lead	0	0	0	0	2.77E-05	4.38E-05	2.49E-06	7.40E-05	1.72E-05	2.57E-05	1.96E-06	4.48E-05

Table A-I-13. Results of Modified CAPCOA Exposure Assessment Model - Estimation of Exposure through Ingestion of Backyard Produce.

Chemical	Leafy 13-20	Vine 13-20	Root 13-20	Total 13-20	Leafy 21+	Vine 21+	Root 21+	Total 21+	**LADD (ug/kg/d)**
Chromium	1.71E-06	2.89E-06	8.62E-08	4.69E-06	3.24E-08	1.99E-07	7.20E-08	3.03E-07	8.14E-06
Benzene	1.81E-08	2.84E-08		4.64E-08					2.26E-08
Cadmium	2.32E-06	5.51E-06	1.65E-06	9.48E-06	1.16E-06	3.17E-06	1.38E-06	5.71E-06	1.93E-05
Vinyl Chloride	4.55E-12	7.14E-12		1.17E-11					5.68E-12
Arsenic	2.76E-07	4.42E-07	2.52E-08	7.43E-07	2.37E-08	4.36E-08	2.11E-08	8.84E-08	3.75E-07
Dioxins	6.74E-11	2.97E-10	4.57E-11	4.10E-10	3.21E-12	2.62E-11	5.70E-12	3.51E-11	2.07E-10
Nitrogen Oxides									
Silicon									
Hydrogen Chloride									
Mercury	4.37E-04	1.05E-03	1.07E-04	1.59E-03	6.27E-05	1.71E-04	2.47E-05	2.58E-04	2.85E-03
Sulphur Dioxide									
Iron	2.10E-04	4.98E-04	1.49E-04	8.57E-04	1.05E-04	2.87E-04	1.24E-04	5.16E-04	1.75E-03
Tin	1.58E-05	3.74E-05	1.12E-05	6.44E-05	7.90E-06	2.15E-05	9.35E-06	3.88E-05	1.31E-04
Particulate emissions									
Lead	8.83E-06	1.40E-05	7.86E-07	2.36E-05	9.24E-07	1.51E-06	6.56E-07	3.09E-06	4.18E-05

Table A-I-14.
Results of Modified CAPCOA Exposure Assessment Model - Summary of Exposure Estimates through All Selected Exposure Pathways

Chemical	LADD					
	Inhalation (ug/kg/d)	Dermal Absorption (ug/kg/d)	Soil Ingestion (ug/kg/d)	Food (ug/kg/d)	Total Ingestion (ug/kg/d)	Ingestion+ Dermal (ug/kg/d)
Chromium	1.91E-06	1.07E-07	9.08E-07	8.14E-06	9.05E-06	9.16E-06
Benzene	3.19E-04	7.51E-08	2.33E-07	2.26E-08	2.55E-07	3.30E-07
Cadmium	9.14E-07	1.02E-08	4.34E-07	1.93E-05	1.98E-05	1.98E-05
Vinyl Chloride	5.13E-06	1.31E-08	4.07E-08	5.68E-12	4.07E-08	5.38E-08
Arsenic	2.80E-07	4.47E-10	3.79E-08	3.75E-07	4.13E-07	4.14E-07
Dioxins	1.04E-10	1.72E-12	3.39E-12	2.07E-10	2.10E-10	2.12E-10
Nitrogen Oxides	8.57E-01					
Silicon	5.33E-02					
Hydrogen Chloride	3.55E-02					
Mercury	5.67E-04	5.55E-06	5.01E-05	2.85E-03	2.90E-03	2.91E-03
Sulphur Dioxide	4.72E-02					
Iron	2.89E-04	9.25E-07	1.12E-05	1.75E-03	1.76E-03	1.76E-03
Tin	2.17E-05	6.95E-08	8.42E-07	1.31E-04	1.32E-04	1.32E-04
Particulate emissions	5.29E-03					
Lead	3.05E-05	4.87E-08	4.14E-06	4.18E-05	4.60E-05	4.60E-05

APPENDIX A-II:

Modelling of Emission Rates, Air Concentrations and Population Risk due to an Incinerator

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A-II-1.0 MODELLING EMISSION RATES AND AIR CONCENTRATIONS DUE TO AN INCINERATOR

Two incinerators were chosen as representative modern incinerators for use in this assessment. The facilities chosen were the Peel Incinerator (KMS Peel Inc.) located near Pearson International Airport and an incinerator in Hartford Connecticut. The Peel incinerator has a waste handling capacity of 400 tonnes/day while the facility in Hartford has a capacity of 2000 tonnes/day.

Emission rates for both facilities were derived assuming the incinerators were operating at capacity. Dispersion modelling of the emissions from the facilities was performed using an emission rate of 1 g/s with concentrations calculated at receptors out to 10 km from the facilities. Concentrations for any compound are derived by multiplying by the emission rate of that compound. Dry deposition of particles released from the stack is then calculated using a deposision velocity of 0.2 cm/s. Wet scavenging depends on the scavenging rate, the frequency of precipitation and the precipitation rate.

A-II-1.1 Incinerator Emissions

Emission rates in g/s for the Peel Incinerator are given in Table A-II-1. These were obtained using stack test data for 1994 given in a report by Bovar-Concord assuming the incinerator was operating at capacity. The maximum emission rates for the incinerator in Hartford Connecticut are also provided in Table A-II-1 (Environment Canada/U.S. EPA, 1991). Since the stack sampling protocols were not the same for the two incinerators, some contaminants occur on only one of the lists.

To perform dispersion model calculations stack characteristics are also needed. To provide a conservative estimate of concentrations due to the facilities, the lowest tested stack exit velocities and temperatures were used in the analyses. The data used for the Peel Incinerator with all 4 units operating was:

Stack Height:	36.9 m
Stack Diameter:	1.67 m
Exit Velocity:	18.2 m/s
Gas Temperature:	159 °C

For the Hartford Connecticut facility with all 3 units operating the stack characteristics were:

Stack Height:	66.5 m
Stack Diameter:	5.10 m
Exit Velocity:	9.5 m/s
Gas Temperature:	120 °C

Note that the stack diameter for the Hartford Incinerator is the effective value corresponding to 3 flues each of which has a diameter of 2.95 m.

A-II-1.2 Other Modelling Data Files

Since annual average air concentrations are one of the outputs required from the dispersion modelling, model runs were performed using a year long hourly meteorological data set. Other than meteorology, emission files and stack characteristics, the other data set needed to perform dispersion calculation is a set of receptor locations at which concentrations are calculated.

A-II-1.2.1 Meteorological Data

To run a dispersion model on an sequential hour by hour basis, meteorological information is needed on the 10 m wind speed and direction, the air temperature, the fractional cloud coverage or opacity and the depth of the convective mixing layer where appropriate. In addition information on ground snow coverage and on the roughness of the ground surface is required by the modelling package used in this study. The information on wind speed and direction as well as air temperature can be obtained for most airports and at other monitoring networks including those maintained by the Ministry. Hourly cloud coverage over the entire day is however only available at the larger airport which are operated throughout the day. To obtain convective mixed layer heights temperature profile data obtained from upper air instrumented balloon releases are needed. This information is only available at specific airports or special sites in Canada and the US.

Wind and temperature data from Buttonville Airport located north of Metropolitan Toronto were selected as the most applicable to this assessment. However, since Buttonville does not have nighttime cloud coverage, information was obtained from Toronto International Airport. Snow coverage was based on daily maps giving the 1 and 6 inch snowlines across North America. These data were combined with convective mixed layer heights obtained using surface temperatures and the morning upper air sondes released at Buffalo. The methodology described in Holtzworth (1967) was used to calculate the daytime mixing heights. The meteorological data set assembled was for one year (1989).

Figure A-II-1 show the wind rose and the wind speed frequency distribution for the data set. The most frequent wind directions are from the north north west and from the west south west.

A-II-1.2.2 Receptor Locations

Calculations were performed for nested grids extending out to 10 km from the facility. Figure A-II-2 shows the locations of the model grid points. A grid resolution of 250 m was used out to

2 km with a 500 m resolution from 2 to 4 km and a 2000 m resolution from 4 to 10 km. The total numder of model receptors was 593.

A-II-1.3 Modelling Method

The dispersion model used in this study is the CAP model developed by MOEE (Appendix 8, Clean Air Program(CAP)). The CAP modelling package consists of different modules for use in specific meteorological conditions. The general categories of meteorological conditions covered by the CAP model are stable (i.e., usually clear skies at night with light winds), neutral (windy and cloudy), convective (daytime with thermal up and down drafts created by the sun heating the earth surface) and lakeshore fumigation (during warm season days near large lakes when a lakebreeze transports stable air from over the lake to the convective airmass over the land). In this modelling package there is a continuous variation of stability as opposed to discrete stability classes. Mechanical mixing is generated by the wind flowing over the surface. The mixing rates depend on the surface roughness. A value of 20 cm was used for the surface roughness in this study.

To evaluate both chronic and acute impacts due to emissions from the incinerator, concentrations were needed over the model domain for the annual average, the maximum 24 hour average and the maximum hour average. All three concentration averages were obtained at each model receptor location. The annual average concentrations depend significantly on the wind rose while the maximum hourly average would be more uniformly distributed with respect to wind direction.

Since air concentrations are directly proportional to the emission rate for single stacks, the calculations were performed with an emission rate of 1 g/s. Concentrations for any compound can be derived by multiplying by the emission rate of that compound.

For particulate emissions from the incinerators, dry deposition and wet scavenging rates were also estimated. The dry deposition rate is simply the ground level air concentration times a deposition velocity. A value of 0.2 cm/s was used as the dry deposition velocity for particulates. To estimate the wet scavenging of particles, information on the wet scavenging rate, the frequency of precipitation and the average precipitation rate was required. Using data from Pearson International Airport, about 900 hours of precipitation occur each year (i.e., precipitation rates above trace amounts) with an average intensity of 1 mm/hour. The formula to determine the fraction of the mass of the plume washed out as a function of distance from the stack is given below:

$$f = \exp(-\mu t) = \exp(-\mu x/u)$$

where μ is the wet scavenging rate (hr^{-1})
t is time in hours
x is distance in km

and u is the wind speed in km/hr (an average value of 18 km/hr was used)

$\mu = aJ$

where a is a scavenging coefficient (mm^{-1})
and J is the precipitation intensity in mm/hr

Measurements of heavy metal concentrations in the INCO plume gave wet scavenging rates in the range from 0.2 to 0.8 hr^{-1} for most compounds. A value of $\mu = 0.5\ hr^{-1}$ was used in this study.

The fraction of the yearly emissions of a compound deposited within a given distance of the incinerator was then determined by multiplying f by the fraction of the hours in a year for which there was more than trace precipitation rates.

A-II-1.4 Results

Figure A-II-3 shows contours of annual average concentration patterns for 1 g/s of emissions from the Peel Incinerator. Over the 10 km region, modelled annual average concentrations distinctly show the predominant north north west wind direction in the meteorological data set. The maximum concentrations are about 1 km from the facility. The lowest annual average concentrations, located in the north west and south west corners of the model domain, are about three orders of magnitude smaller than the concentration maximum at the model boundary.

Figures A-II-4 and A-II-5 depict the maximum 24 hour average and the maximum hour average concentrations respectively. These maxima were selected from the year long meteorological data set at each individual receptor location. The plot for 24 hour average maxima still shows that the wind direction frequency plays a part in the distribution of concentrations although not quite as much as for the annual average. However, the maximum hourly average concentration occurs when the worst dispersion conditions prevail (light to moderate wind speeds and unstable conditions are worst case for these stack emissions). These conditions can occur with any wind direction but will occur more frequently for the predominant wind directions. The highest 24 hour average concentration was about 23 times as large as the highest calculated annual average concentration. For the hourly average, the highest value was about 68 times the highest calculated annual average concentration. To derive concentrations for specific compounds, the results in Figures A-II-3, A-II-4 and A-II-5 are simply multiplied by the emission rate of the compound of concern.

Similarly Figures A-II-6, A-II-7 and A-II-8 are respectively the annual average, the maximum 24 hour average and the maximum hourly average for the Hartford Connecticut Incinerator. Because this is a taller stack with more buoyancy flux to promote plume rise, concentrations due to 1 g/s of emissions are much lower than found for the Peel Incinerator. The annual average concentration for 1 g/s of emissions is nearly a factor of 7 lower than the Peel Incinerator result.

The highest 24 hour average concentration was about 24 times the largest annual average value at this facility while the highest hourly average value was about 110 times the annual average maximum.

GLC maxima and the location of POI for both the Peel and Hartford facilities are shown in table 2.

A-II-2.0 MODELLING OF POPULATION RISK

Calculating the number of expected cases of cancer due to emissions from an incinerator requires the following steps.

- The emission rates for each compound are required to estimate air concentrations around the facility. Stack characteristics are also required to perform dispersion modelling. The information used in this study is described in Appendix A-I.
- A dispersion model is then run using a year long hourly meteorological data set to calculate annual average concentrations over a 20 km by 20 km area around the incinerator. See Appendix A-1 for a description of the methodology used to calculate concentrations due to incinerator emissions.
- The risk per $\mu g/m^3$ for each compound for all appropriate exposure pathways must first be obtained. The procedures to estimate risks and the uncertainty in those risks are described below.
- Information on the best estimate and the uncertainty in each of the above factors is then used to estimate risk distributions for each compound at the point of maximum concentrations or at any other location around the incinerator. This is done for all compounds of concern.

To estimate the maximum likely population risks around an incinerator, the facility is assumed to be surrounded by a large urban population. For consistency between the landfill and incineration studies, the incinerator was assumed to have the population around the Britannia Landfill in Mississauga. About 550,000 people live within 10 km of this site. Since the population distribution around a site can vary and the patterns of concentrations for a landfill and an incinerator are different, the fairest estimation of the relative population risks due to the two operations requires that the population distribution be treated statistically. The steps in calculating a population risk along with an uncertainty estimate which depends on where the population is located relative to the facility are:

- Use census tract data for the area within 10 km of the site to derive a distribution of population densities averaged over 2 km by 2 km grid cells. Given the very large population around this landfill, this provides a maximum likely population risk. Figure

B-6 in the accompanying report - *Report B: Assessment of the Potential Health Risks Associated with Chemical Emissions from Municipal Solid Waste Landfills* shows the bimodal distibution of population densities derived from the census tract data. The low density mode corresponds to mainly industrial or mainly rural areas while the high density mode represents various types of residential development.

- The above population density data is used to create a customized distribution curve using Crystal Ball Version 3.0 (Decisioneering, Inc., Denver, CO).

- Air concentrations for a 1 g/s emission rate are derived as 2 km square averages over a 20 km by 20 km area around the landfill. Since the pattern of concentrations are the same for all compounds in this study, calculations for individual compounds can be derived by multiplying by that compound's emission rate in g/s.

- Using a Monte Carlo sampling procedure, a population density is selected from the distribution curve for each of the 100 grid cells for which concentration data is available. The selected population in each cell is multiplied by that cells concentration and the results are then summed over the 20 km by 20 km domain. The final result is a value for population weighted exposure to concentrations of a given compound (i.e., units are Population*$\mu g/m^3$).

- The above calculation can then be repeated for many random selections (≈100 cases were used) of population densities around the landfill resulting in a series of possible population exposures to concentrations. Figure A-II-9 shows the results of this analysis technique for population exposure to chromium (Cr^{6+}) after multiplying by the emission rate from the Peel Incinerator.

- Multiplying the individual risk per $\mu g/m^3$ for each appropriate exposure pathway for the compound by the population concentration exposure gives the number of people affected over the 30 year period. Since both of these factors have a distribution associated with them, the final answer would be a distribution for the number of people affected. This distribution would represent the combination of all of the uncertainties and variabilities in the overall assessment. This includes the uncertainties in the risk estimates for the compound, variations in emission estimates from incinerator facilities, and the variation due to the location of population centres relative to the incinerator location.

- Having derived population risk numbers for one compound, the risks for other compounds are readily calculated since the risks all vary linearly with concentration and the pattern of concentrations is the same for all compounds.

A-II-3.0
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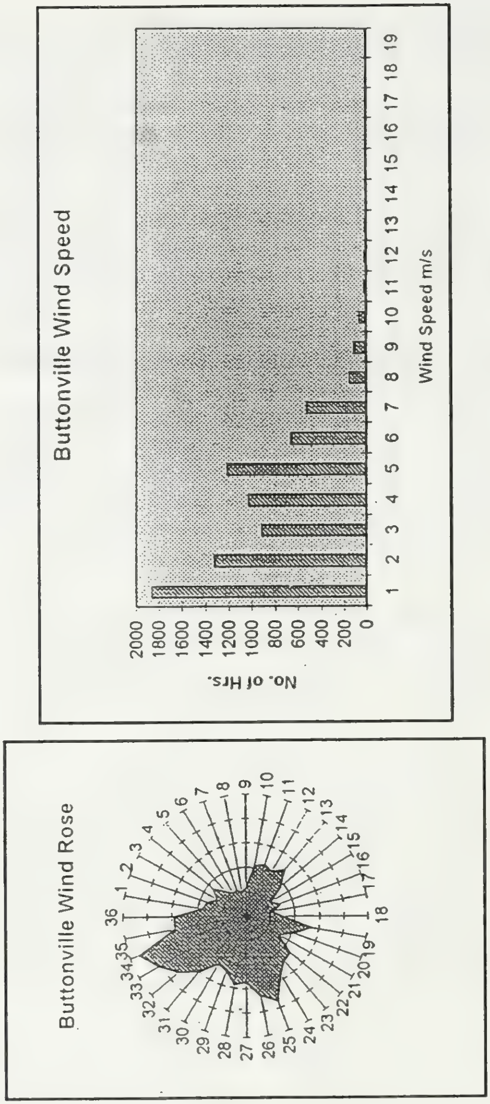

Figure A-II-1Wind rose and wind speed frequency distribution for Buttonville Airport.

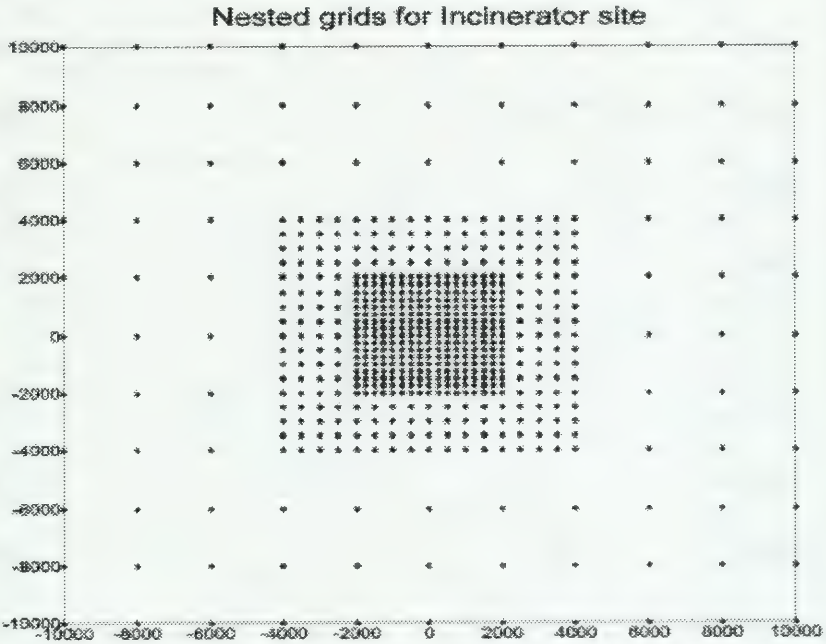

Figure 2: Model receptor locations for a 20 km by 20km grid

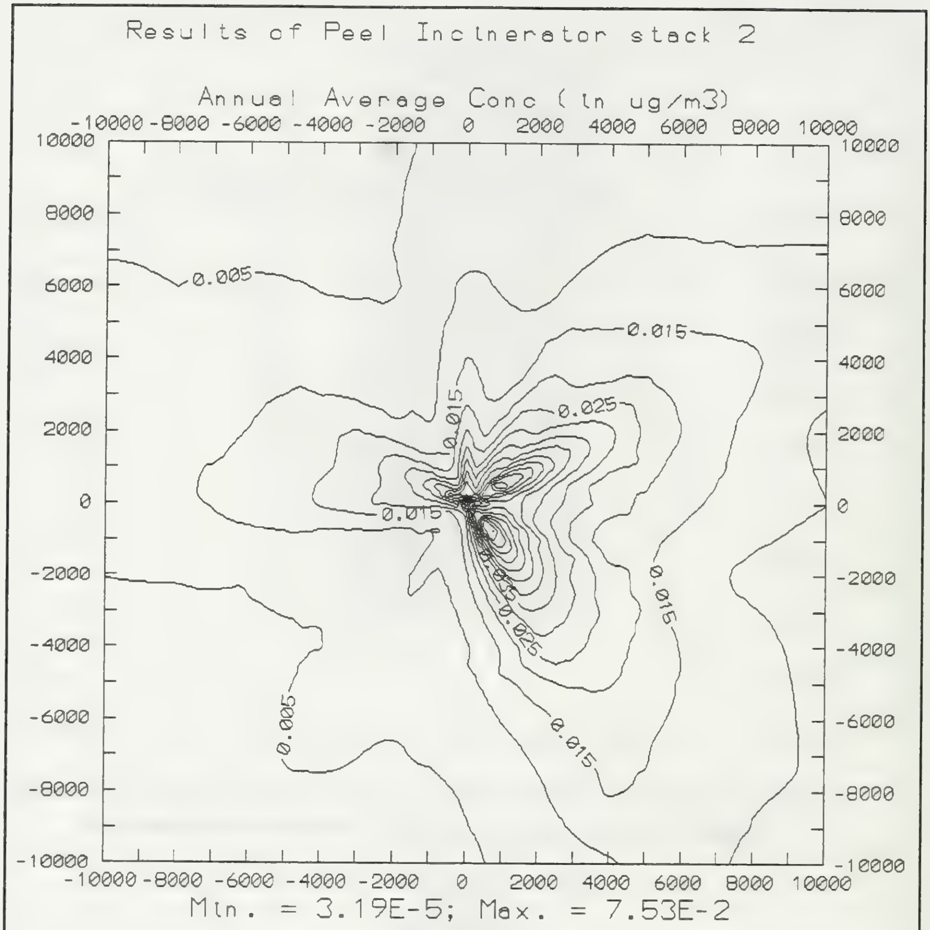

Figure 3: Modelled annual average concentrations for Peel Incinerator (1g/s).

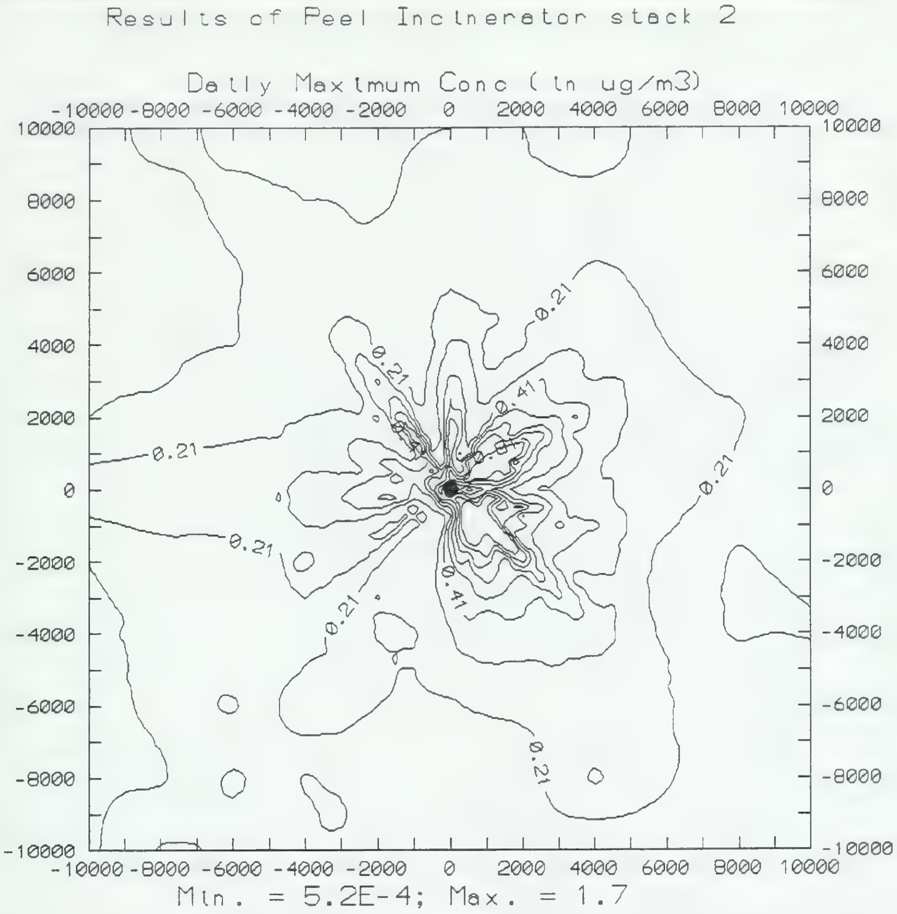

Figure 4: Modelled maximum 24 hour average concentrations for Peel Incinerator (1g/s).

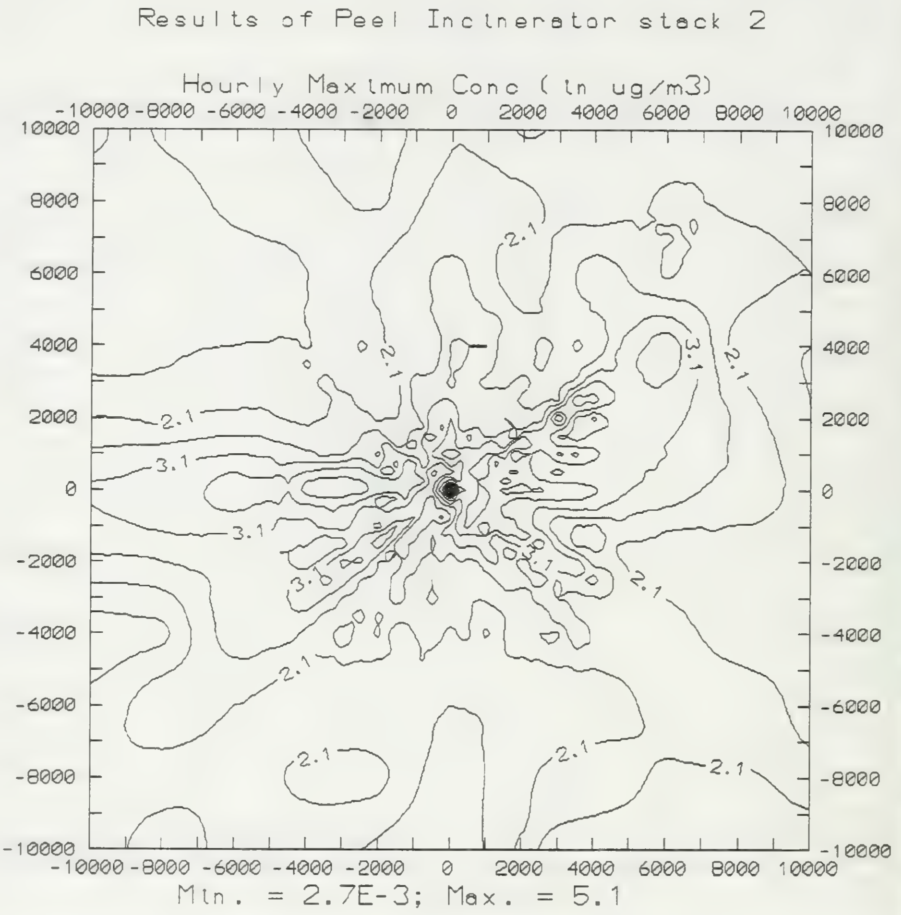

Figure 5: Modelled maximum 1 hour average concentrations for Peel Incinerator (1g/s).

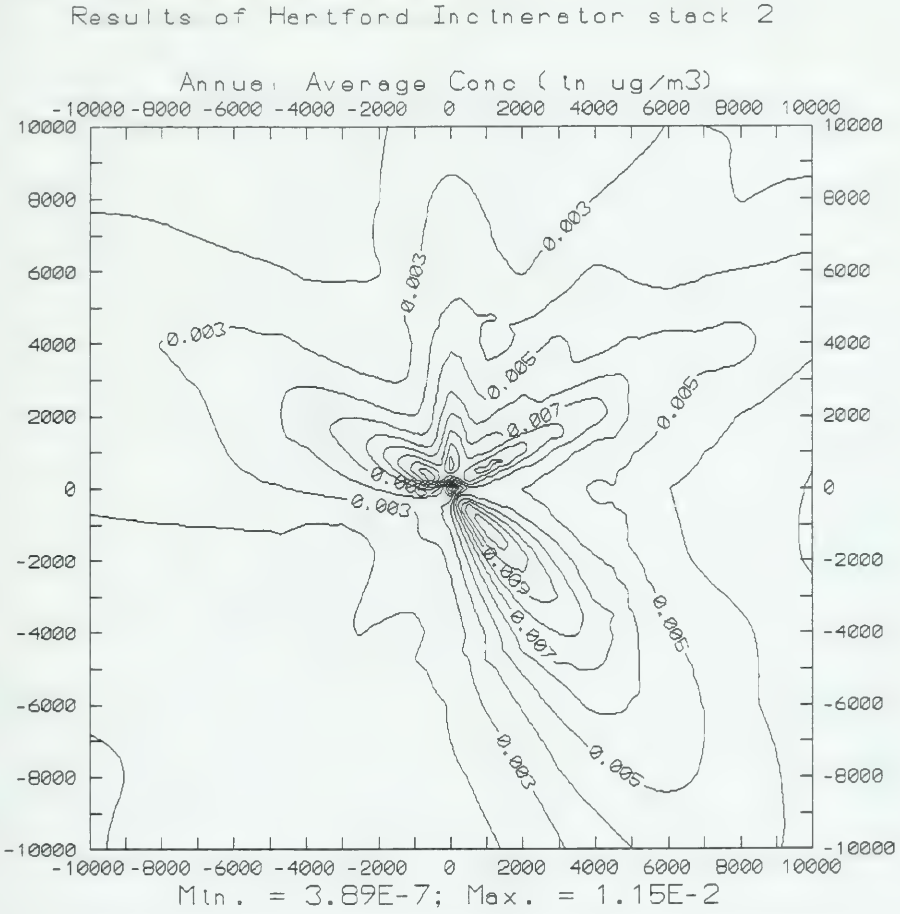

Figure 6: Modelled annual average concentrations for Hardford Incinerator (1g/s).

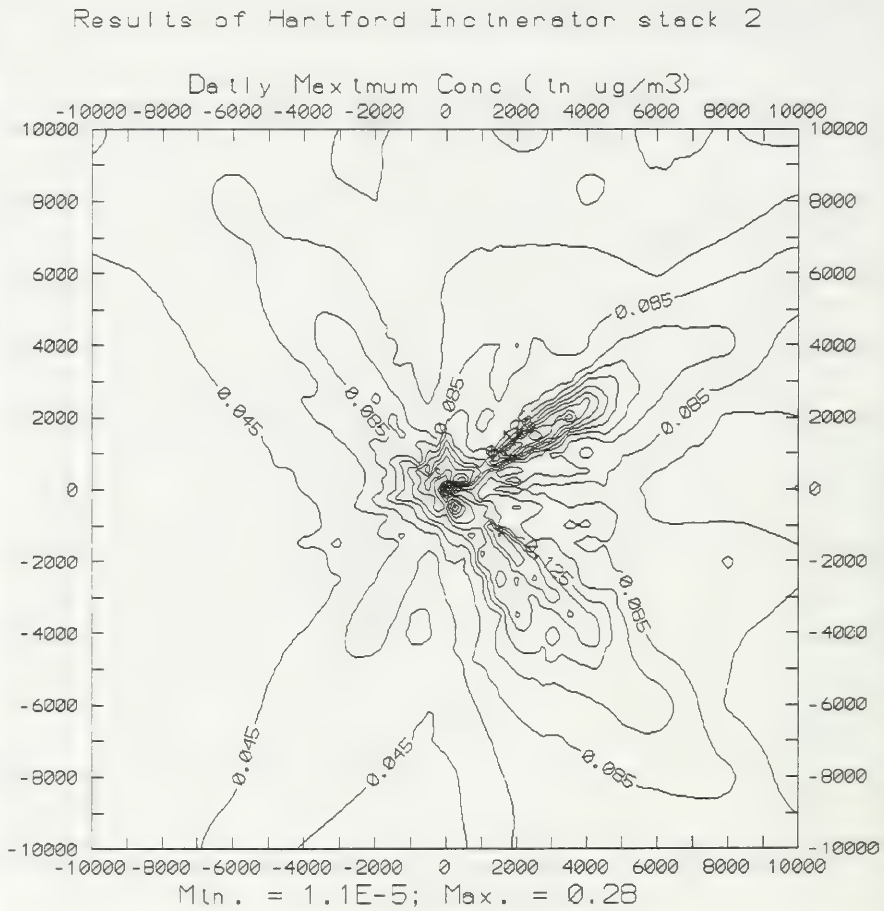

Figure 7: Modelled maximum 24 hour average concentrations for Hardford Incinerator (1g/s).

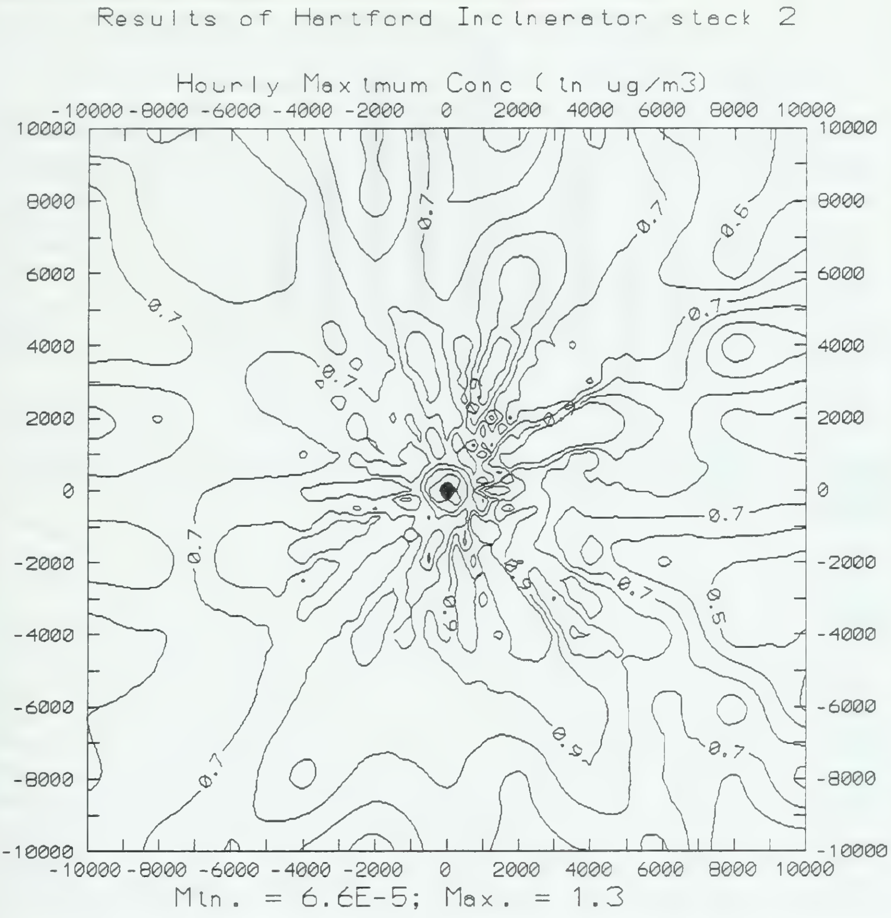

Figure 8: Modelled maximum 1 hour average concentrations for Hardford Incinerator (1g/s).

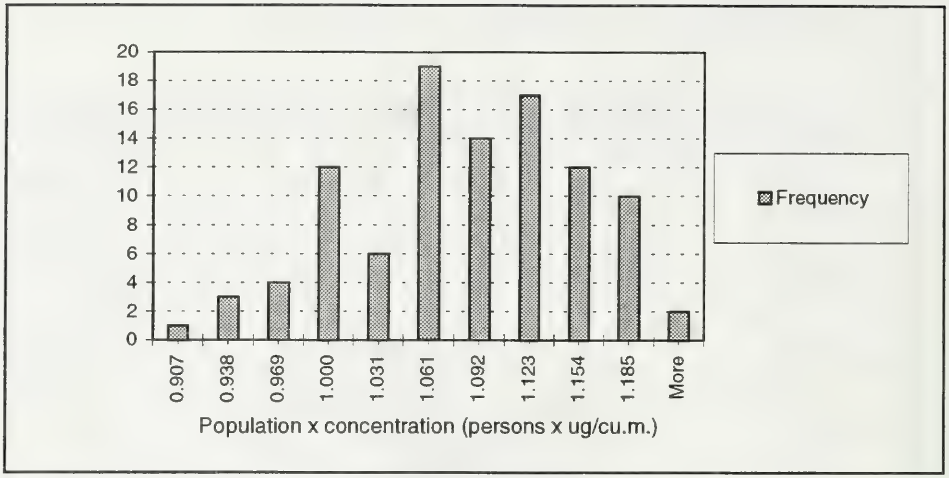

Figure A-II-9. Population-weighted Concentration for Chromium emissions from a MSW Incinerator.

Table A-II-1 Emission rates for MSW Incinerators used in this study.

	Hartford[1]	Peel[2]
Stack Testing Parameters	Corrected Emission rate (ug/s)	Corrected Emission rate (ug/s)
Particulate emissions	821083.02	117739.88
Sulphur Dioxide	821083.02	1051143.83
Nitrogen Oxides	991797705.00	19044242.00
Hydrogen Chloride	12007980.00	790781.72
Aluminum	1093.09	1093.09
Antimony	42.15	42.15
Arsenic	21.80	21.80
Barium	58.14	58.14
Beryllium	9.16	9.16
Boron	2944.95	2944.95
Cadmium	71.23	71.23
Calcium	52894.28	52894.28
Chromium	2162.93	148.56
Cobalt	159.31	159.31
Copper	240.86	240.86
Iron	6440.81	6440.81
Lead	7182.13	678.82
Lithium	127.62	127.62
Magnesium	1000.06	1000.06
Manganese	293.62	293.62
Mercury	12631.60	12631.60
Molybdenum	117.16	117.16
Nickel	2395.50	2395.50
Phosphorus	1266.07	1266.07
Selenium	52.33	52.33
Silicon	1187256.94	1187256.94
Silver	92.16	92.16
Sodium	21191.72	21191.72
Strontium	30.53	30.53
Tin	483.90	483.90
Titanium	104.66	104.66
Vanadium	31.98	31.98
Zinc	5487.26	5487.26
Fluoride	21.22	21.22
1,2-Dichlorobenzene	56.25	2.05
1,3-Dichlorobenzene	17.72	1.93

1,4-Dichlorobenzene	23.69	23.69
1,2,3-Trichlorobenzene		1.56
1,2,4-Trichlorobenzene		3.88
Total Trichlorobenzenes	26.46	5.44
1,2,3,4-Tetrachlorobenzene		1.67
1,2,3,5- & 1,2,4,5-Tetrachlorobenzene		3.92
Total Tetrachlorobenzenes	6.11	5.59
Pentachlorobenzene	3.72	3.72
Hexachlorobenzene	0.41	0.41
2,5-2,4-Dichlorophenol		5.60
2,6-Dichlorophenol		1.67
Total Dichlorophenols	41.86	7.27
2,3,4-Trichlorophenol		3.91
2,4,5-Trichlorophenol		1.83
2,4,6-Trichlorophenol		7.70
Total Trichlorophenols	26.46	13.44
2,3,4,6-Tetrachlorophenol	4.78	4.78
Pentachlorophenol	5.67	2.76
Tetralin	13.71	13.71
Biphenyl	82.03	82.03
o-Terphenyl	2.25	2.25
2-Methylnaphthalene	7.56	7.56
1-Methylnaphthalene	2.70	2.70
2-Methylfluorene	420.08	420.08
1-Methylpyrene	0.47	0.47
Naphthalene	38.55	38.55
Acenaphthylene	5.67	0.77
Acenaphthene	2.62	2.62
Fluorene	3.20	1.31
Phenanthrene	16.86	6.24
Fluoranthene	19.19	1.10
Pyrene	10.47	1.03
Benzo(b)fluoranthene	0.44	0.15
Benzo(a)fluorene	0.76	0.76
Benzo(b)fluorene	0.52	0.52
Total I-TEQ	0.01	0.02
Total Chlorobiphenyl (PCBs)	1.88	1.88
Chloromethane	4321.49	1440.50

Bromomethane	989.89	329.96
Dibromoethane	11.92	3.92
Dichlorodifluoromethane	2394.04	710.80
Trichlorofluoromethane	4735.76	1578.59
Vinyl Chloride	1199.20	399.73
1,1-Dichloroethene	15.55	5.23
Trichloroethene	13.52	4.50
1,1,1-Trichloroethane	39.25	13.08
Tetrachloroethene	62.21	20.70
Benzene	74664.53	24889.63
Toluene	1382.35	463.69
Ethylbenzene	28.49	9.45
Total Xylenes	16610.04	5538.13

1. Peel/Hartford MSW Incinerator - Maxima from 1993 & 1994 Peel tests OR 3X average of 1 boiler tests from Hartford
All values corrected for stack temp to ground level (25 Celsius)

2. Peel MSW Incinerator - Maxima from 1993 & 1994 tests + positives (average of 1 boiler tests) from Hartford for some Peel ND
All values corrected for stack temp to ground level (25 Celsius)

Table A-II-2. Results of Dispersion Modelling

Source Name (emitting @ 1 g/s)	Maximum Ground Level Concentration (ug/m^3)	Maximum Point of Impingement Location (m)
Peel - Hourly	5.1×10^{-6}	5831 NE
Peel - daily	1.7×10^{-6}	901 NE
Peel - annual average	7.5×10^{-8}	1060 SE
Hartford - hourly	1.3×10^{-6}	3354 NE
Hartford - daily	2.8×10^{-7}	2916 NE
Hartford - annual average	1.15×10^{-8}	1414 SE

APPENDIX A-III:Distributions for Exposure Variables Used in MWSI Probabilistic Analysis

Appendix A-III. Distributions for Exposure Variables Used in Probabilisitc Analysis.

Parameter	Distribution	Mean or Likeliest	Standard Deviation	Minimum	Maximum	Reference
Surface Area (Proportion)						
SA - Arms	T	0.14		0.12	0.15	
SA - Hands	T	0.05		0.05	0.05	
SA - Legs	T	0.21		0.18	0.23	
Total SA (cm^2)/ kg	N	641	114	- ∞	+ ∞	
Emissions						
Benzene	U	10.5		5	16	
Vinyl Chloride	U	105		30	183	
Kow Vinyl Chloride	T	14		4	25	
Kow Benzene	T	135		36	490	
Koc Vinyl Chloride	T	32		3	57	
Koc Benzene	T	83		12	339	
Chromium	LN	1.12E-05	3.92E-05	- ∞	+ ∞	
Benzene	LN	1.87E-03	6.55E-03	- ∞	+ ∞	
Cadmium	LN	5.34E-06	1.87E-05	- ∞	+ ∞	
Vinyl Chloride	LN	3.00E-05	1.05E-04	- ∞	+ ∞	

Arsenic	LN	1.64E-06	5.74E-05	- ∞	+ ∞	
Dioxin	LN	6.11E-10	2.40E-09	- ∞	+ ∞	
Nitrogen Oxides	LN	2.86	10	- ∞	+ ∞	
Silicon	LN	8.9E-02	3.1E-01	- ∞	+ ∞	
Hydrogen Chloride	LN	5.93E-02	2.1E-01	- ∞	+ ∞	
Mercury	LN	9.47E-04	3.3E-03	- ∞	+ ∞	
Sulphur Dioxide	LN	7.88E-02	2.8E-01	- ∞	+ ∞	
Iron	LN	4.83E-04	1.7E-03	- ∞	+ ∞	
Tin	LN	3.63E-05	1.3E-04	- ∞	+ ∞	
Particulates	LN	8.83E-03	3.1E-02	- ∞	+ ∞	
Lead	LN	5.09E-05	1.8E-04	- ∞	+ ∞	

LN: log-normal distribution; U: uniform distribution; T: triangular distribution; N: normal distribution

TECHNICAL REPORT – PART "B"

ASSESSMENT OF THE POTENTIAL HEALTH RISKS ASSOCIATED WITH CHEMICAL EMISSIONS FROM MODERN MUNICIPAL SOLID WASTE LANDFILLS IN ONTARIO

EXECUTIVE SUMMARY

A predictive toxicological risk assessment was conducted in order to evaluate the occurrence of potential human health effects associated with the emissions of a hypothetical municipal solid waste landfill (MSWL). This analysis is part of a more detailed comparative assessment between the projected health and ecological impact associated with the MSWL and those from a typical municipal solid waste incinerator (MSWI).

The MSWL considered for this analysis was defined based on state-of-the-art technological and scientific criteriaconsidered by the Ontario Ministry of the Environment prior to the release of the 1998 Landfill Standards (OMOE, 1998) . These criteria include, among others, considerations for the choice of an appropriate location based on variables such as 1) projected capacity and operational lifespan of the landfill, 2) existing and projected human populations, 3) natural ecosystems, 4) geomorphological and hydrogeological factors, and 5) state-of-the-art capacities for the collection and treatment of the leachate, and for the collection and flaring of gaseous and volatile compounds generated as a result of the aerobic and anaerobic digestion of municipal solid wastes. Because of the complex interdependence of these parameters and time and resources constraints, these variables were not explored in detail in the present analysis. The methodology adopted, instead, consisted in reviewing the information available on actual and proposed sites in Ontario, and adopting one for which these considerations have been reviewed. This methodological choice was dictated strictly by practical considerations and does not bear any implication for the site(s) in the foreseeable future.

In order to provide a realistic basis for this comparative assessment, both the MSWL and MSWI were assumed to be located at the same geographical location representing a typical suburban area. This ensured that both facilities would experience identical meteorological conditions (*e.g.*, dominant winds) and impact upon the same populations and ecosystems. It was felt that this would also provide comparable grounds for estimating the movement of chemicals in various media, their subsequent intake by pre-defined human, aquatic, and terrestrial receptors and, thus, provide a common basis for the computation and comparison of the estimated risks associated with the emissions from both facilities.

The MSWL health risk assessment attempted to estimate potential health risks associated with atmospheric emissions of carcinogenic and systemic toxicants; nuisance problems associated with the atmospheric emissions of odorous compounds; and the potential health impact of carcinogenic and systemic toxicants present in the raw leachate that may leak into, and be transported by, the underlying aquifer to target receptors. Although it has been recognized that other ailments and more subtle ill-defined syndromes (*e.g.*, headache, nausea) may be associated with landfill emissions, their characterization remains hypothetical within a predictive framework in view of the unknown etiological agents associated with these effects and the underlying mechanisms of action involved. Consequently, these aspects were not examined in detail in the present assessment. Furthermore, the potential health and comfort impact of particulate released during the operational phase of the landfill were not addressed, although this might be considered in any future refinement envisioned for this analysis.

These aspects were investigated with two conceptually different risk assessment approaches. The first examined individual exposures and the ensuing potential health and discomfort effects for a series of human receptors living at the location corresponding to the maximum ground-level concentration (GLC) of emitted chemicals (*i.e.*, point of maximum impact). This is termed a deterministic or "point" or reasonable worst case estimate. In the case of chemicals associated with chronic adverse effects (*e.g.*, carcinogens, systemic toxicants) this deterministic analysis was extended to incorporate a probabilistic

analysis in order to obtain ranges of individual exposures. The second approach also examined, probabilistically, the potential health risks and effects for a population of approximately 550,000 living in a 20 km x 20 km perimeter surrounding the facility. This component was essential in order to characterize the extent of the impact of each facility on a broader scale and to provide a more realistic basis for comparing health risks in view of the different geographical locations associated with the maximum GLC for the landfill (near the property line) and the incinerator (within one to several kilometres downwind). This assessment contains 3 major phases, namely 1) screening, 2) modelling of concentrations in different environmental media, and 3) characterization of potential health risks and effects.

In phase one (Screening), chemicals were initially classified according to two major categories of health effects, namely carcinogenicity and systemic (or non-carcinogenic) effects; these were further subdivided into more specific classes of effects based on similarities in end-points/target organs and on the assumed nature of the mechanisms involved. In this regard, the categories retained for the carcinogens included *chlorinated organics* and *others* whereas systemic toxicants were examined based on their individual effects as well as for their capacity to induce reproductive and developmental effects and fetotoxicity. Compounds associated with odor nuisance were also examined. Screening was then performed by estimating the lifetime excess cancer risk (LECR) (for carcinogens) and exposure ratios (ER)(for systemic toxicants and odorous compounds) under worst case conditions. In the case of the atmospheric emission components, a final set of 11 carcinogens, 6 systemic toxicants (including 3 heavy metals), and 6 chemicals associated with odor issues were retained from an initial list of 63 chemicals. In the case of the leachate, one chemical was retained from an initial list of 169 chemicals.

In the second phase, the levels of chemicals retained from the previous screening step were initially modelled over a predefined area surrounding the facility (20 km x 20 km) in order to generate 1-hour, 24-hour, and annual average atmospheric ground-level concentrations for different time windows corresponding to the operational (0-4, 5-8, 9-12, 13-16, and 17-20 years) and post-closure (21-25, 26-30, 31-40 years) phases of the landfill. In the case of the leachate, modelling results were used in order to estimate the nature and amounts of chemicals migrating to the aquifer located underneath the receptors following diffusion through landfill protective membranes and transport in the groundwater. A multimedia exposure algorithm was then used in combination with the atmospheric modelling results to examine the contribution from various environmental pathways and routes of exposure to the total intake of chemicals. This algorithm was used both deterministically using reasonable worst case assumptions and then extended to incorporate a probabilistic analysis. These included 1) direct intake by inhalation, 2) intake following ingestion of soil and garden produce contaminated by depositions and/or air transfer of airborne emissions, and 3) intake following dermal contact with contaminated soil. The choice of these pathways was dictated by the location selected which reflects a suburban area with the possibility of demographic expansion in the perimeter surrounding both the landfill and the incinerator, a situation frequently observed in southern Ontario. This choice, however, precluded the assessment of pathways and/or routes that would have been considered relevant under different circumstances (*i.e.,* those associated with agricultural activities and the consumption of sport fish). The potential health impact of contaminated groundwater at the receptors location was examined by assuming direct ingestion. This is a worst case assumption which only applies to residents using well water. No other pathways and routes of exposure were considered for this component as it is expected that the majority of sub-urban residents will be served by municipal treatment plants from their or nearby communities. Indirect pathways such as vapour diffusion through soil and basement infiltration were not considered in the present analysis.

In the third phase of this assessment, health risk estimates were evaluated for a collection of receptors reflecting different stages of life. These include the following age groups: 0-6 months, 7m-4 years, 5y-11y, 12y-19y, and 20+. Specific intake rates and variables reflecting physical characteristics for each of these subgroups were appropriately combined, according to predetermined scenarios, with the concentrations of chemicals modelled in the relevant environmental media in order to obtain risk estimates and exposure ratios for carcinogens and systemic toxicants, respectively. The assessment of potential odor problems was conducted by comparing maximum 10-minute concentrations for selected chemicals with olfactory thresholds reported in the literature or available from Ministry databases. In order to estimate individual risks, the receptors were conservatively assumed to reside for a period of 30 years at the location associated with the maximal average ground-level concentrations (corresponding to the fenceline). Population risks estimates were based on weighted average concentrations of chemicals modelled for the whole 20 km x 20 km area.

Results obtained for the chlorinated class of carcinogenic compounds present in atmospheric emissions (obtained by applying a deterministic approach based on measures of central-tendency) indicated that the LECR for individual compounds were all less than or around the accepted regulatory risk level of 1E-06. Only vinyl chloride slightly exceeded this mark (1.6E-06, respectively). When total LECR for this class of compounds was estimated under the assumption of additivity of effects, the value obtained was approximately 4E-06. In this case, the major contribution to total intake was associated with direct inhalation (≈ 97%), followed by ingestion of soil (≈ 2.2%), dermal contact with soil (≈ 0.8%), and ingestion of garden produce (≈ 0.05%). A total LECR of 5.1E-08 was estimated for benzene, the only representative selected from the *others* category.

When total LECR were estimated for the group of chlorinated carcinogens using the probabilistic Monte-Carlo-based methodology, the mean LECR was approximately 3.6E-06 whereas LECR associated with the 95th and 97.5th percentile of the resulting distribution were 7.9E-06 and 1.0E-05, respectively. Estimates of LECR obtained for vinyl chloride, a compound of current toxicological concern frequently detected at landfill sites in Ontario were 1.5E-06, 4.2E-06, and 5.7E-06 at the mean, 95th, and 97.5th percentiles, respectively. Deterministic estimates of LECR calculated for the vinyl chloride potentially present in the groundwater aquifer at receptor location indicated insignificant risk (4.0E-10).

Exposure ratios estimated for the selected non-carcinogenic systemic toxicants (hydrogen sulphide, ethyl mercaptan, and methyl mercaptan) were below 1 with the exception of methyl mercaptan which only exceeded one at the 97.5th percentile of the distribution obtained for the 17-20 years period (*i.e.*, period corresponding to maximum emissions). Among the 3 elements selected for the final assessment (iron, zinc, and lead), only iron displayed ER above unity at the 95th and 97.5th percentiles. However, based on various considerations pertaining mainly to the nature of the assumption used to estimate the GLC of particulate-bound metals at receptor location, it was concluded that these exceedances were not toxicologically significant.

Population risks (expressed as the potential number of cases among the whole population) were estimated only for the chlorinated class of carcinogens. In this regard, the highest value obtained (0.6 possible cases) was for the whole mixture at a level corresponding to the 100th percentile. The practical implications suggested by these results imply that there would be no cases of adverse health effects among a population of approximately 550,000 living within a 20 km x 20 km area around the landfill.

Finally, ER greater than unity suggest exceedance of odor thresholds may occur for individuals residing at the point of maximum impact during the assumed 30 years of residence were obtained for ethyl and methyl mercaptan based on modelled maximum 10 minutes GLC. Maximum values attaining ER of 27 (ethyl mercaptan) were estimated for the 13-16 emission period whereas minimum ER were approximately of 1.7 (methyl mercaptan) for the 1-4 period. Additional modelling conducted over the 20 x 20 km area also suggest that odor problems are likely to be experienced by the community.

In summary, the results obtained in this predictive analysis suggest that no adverse health effects (*i.e.*, carcinogenicity and systemic dysfunctions, including reproductive effects) are likely to be experienced by a typical community residing around an existing state-of-the-art MSW landfill. On the other hand, nuisance problems associated with the presence of odorous compounds may constitute a limiting factor for some of these individuals. These projections find support from the practical experience gained by public health and government officials who have witnessed a high reporting of odor complaints from communities residing in proximity of MSW landfills in Ontario. They are also confirmed by several community health studies published in the peer-reviewed scientific literature on hazardous and sanitary waste sites which have reported that the prevalence of overt adverse effects (*e.g.*, cancer, total mortality, and reproductive outcomes) were not higher in the exposed than in the control populations whereas there was generally a significant increase in the complaints associated with odors and subjective symptoms.

The message conveyed by this analysis must be interpreted with caution when attempting to evaluate other scenarios. Indeed, the conclusions obtained were for an existing state-of-the art (prior to the 1998 Landfill Standards) MSW landfill of pre-defined capacity equipped with gas and leachate collection and treatment systems which meet specific criteria of efficiency and reliability. A new municipal waste landfill will have to meet the requirements of Ontario's stricter Landfill Standards guideline. As a result, a new facility would be expected to present even less risk than shown in this analysis.

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B-1. INTRODUCTION AND BACKGROUND

The disposal of municipal solid wastes (MSW) represents an issue of current environmental concern for the municipalities of Ontario. This phenomenon has been amplified during the past decade by the dramatic demographic expansion experienced by numerous urban areas across Southern Ontario, thus creating an immediate need for rapid, efficient, and economical solid waste reduction and disposal programs. Although various approaches are currently available or in the process of refinement, disposal through landfilling and incineration remain the methods of choice of any integrated solid waste management policy (Ham, 1992; O'Leary *et al.*, 1987).

A major concern associated with the operation of landfills and incinerators regards the potential health and nuisance effects that may be associated with agents released from these facilities (Buffler *et al.*, 1985). A wide spectrum of minute amounts of organic compounds, heavy metals, noxious gases, particulate, and bioaerosols may be emitted in the air or released to the soil during the normal operation of these facilities. However, the public health significance of these emissions is largely unknown, partly because the studies available on existing facilities were conducted in order to address perceived or actual acute symptoms and, hence, alleviate transient problems (Deloraine *et al.*, 1995; Lipscomb *et al.*, 1991; Hertzman *et al.*, 1987; US DHUS, 1992), partly because our capacity to predict these effects is limited by insufficient scientific and biomedical information and by the complexity of the issue at hand. The latter aspect is better appreciated when one considers the numerous uncertainties associated with the temporal nature of the emissions, their intensity and frequency; the number and nature of the agents emitted, their fate and transformation in the various environmental media (*e.g.*, air, water, soil, living organisms); the way people may be exposed to these agents; and the ultimate effect(s) that these agents and their decomposition products, individually or in combination, will have on specific target groups presenting with different susceptibilities (*e.g.*, infants, children, elderly, asthmatics, pregnant women, the chronically ills and compromised immunologically).

Certain of these concerns can be explored prospectively with the application of modern risk assessment techniques. These approaches, which borrow from disciplines such as toxicology, environmental sciences, and public health are extensively used today by regulatory agencies and health organizations in order to formulate research priorities, identify potential environmental health concerns, and provide a basis for the development of protective regulations. When used properly, these approaches allow for early recognition of potential health hazards and the circumstances under which these hazards may be expressed, thus permitting the development and implementation of preventive measures. Because of the limitations inherent to any predictive approach, however, certain possible outcomes can only be hypothesized and, consequently, must await further confirmation through field studies. As such, predictive risk assessment must be viewed as constituting a filtering process that help formulate inferences within the limits of what can be predicted based on the available scientific evidence.

It is with these conceptual limitations in mind that the Ontario Ministry of Environment (OMOE) has undertaken a major comparative risk assessment study between the possible health and ecological impact associated with landfilling and incineration of MSW. The primary goal of this investigation was to provide municipalities and other intervenors with baseline information for any prospective risk assessment envisioned for a specific facility. For this purpose, this predictive study examined realistic scenarios regarding the technical aspects associated with state-of-the art facilities as they may be promoted in the province of Ontario, the nature of their emissions, the populations affected, and a spectrum of health effects that may be reasonably addressed within a predictive risk assessment framework.

The present analysis reports on the aspects associated with the potential health effects associated with emissions from MSW or sanitary landfills. The other components pertaining to ecological effects of landfill emissions, and those describing the possible health and ecological effects from MSW incinerators are reported in the accompanying documents. Section B-2 of the present report describes the methodology adopted and the results pertaining to the preparative phases of the assessment. Results are presented in Section B-3, and are further discussed in Section B-4. Individual data, input for the models, quantitative assumptions, and other secondary material are reported in Appendices B-I, B-II and B-III.

B-2. METHODS

B-2.1 Summary of the Methodological Approach

The MSW landfill considered in this study was defined based on existing or proposed state-of-the-art technological and scientific criteria recommended and/or accepted by the Ministry prior to the release of the 1998 Landfill Standards (OMOE, 1998). These criteria include, among others, considerations for the choice of an appropriate location based on variables such as a) projected capacity and operational lifespan of the landfill, b) existing and projected human populations, c) natural ecosystems, d) geomorphological and hydrogeological factors, and e) state-of-the-art capacities for the collection and treatment of the leachate, and for the collection and flaring of gaseous and volatile compounds generated as a result of the aerobic and anaerobic digestion of municipal solid wastes. Because of the complex interdependence of these parameters and time and resources constraints that complicate development of realistic technical scenarios, these variables were not explored in detail in the present analysis. Instead, the strategy adopted consisted in reviewing the information available on actual and proposed sites in Ontario and selecting adequate information for risk modelling. This methodological choice was dictated strictly by practical considerations and was not intended to have any implication for future Ministry policy.

The generic landfill was selected from those submitted for consideration to the former Interim Waste Authority (IWA). Projected site EE11 located in the Municipality of Durham, Ontario, was retained as a surrogate for the present comparative risk assessment based on the technical arguments discussed previously. A summary of the characteristics pertaining to this site are reported in Appendix B-I. Supplementary information can also be found in the series of documents on the subject (IWA, 1994). The site chosen for the study has a capacity of 6.6 million tonnes with a lifespan of 20 years. The site is expected to have a landfill gas (LFG) collection and enclosed flare control system by the end of year 4. The collection system during the active landfilling period will consist of a network of horizontal trenches, supplemented by vertical wells in areas that have reached final contour. Although a comprehensive health risk assessment has been carried out for the projected EE11 facility, several technical and scientific inconsistencies have been identified with the reporting and analysis of toxicological data, the development of exposure limits for a number of chemicals, and with other critical components of the assessment (Manca and Leece, 1995). These concepts and approaches were not retained for the present study although others which, in our opinion, proved relevant and defensible were included. Whenever appropriate these elements are identified in the present study and referred to the original document(s).

The probability that adverse effects may develop in a pre-defined number of people (or the number of possible cases) were estimated for a population living around the hypothetical landfill for a period of 30 years. Demographics associated with the community surrounding the Brittania landfill, located in the municipality of Mississauga, Ontario, were selected for this purpose.

In order to provide a realistic basis for this comparative assessment, both the landfill and incinerator were assumed to be located at the same geographical location. This ensured that both facilities would experience identical meteorological conditions (*e.g.,* dominant winds) and impact upon the same populations and ecosystems. It was believed that this would also provide comparable grounds for estimating the movement of chemicals in various media, their subsequent intake by pre-defined human, aquatic, and terrestrial receptors and, thus, provide a common basis for the computation and comparison of the ultimate risks associated with the emissions from both facilities. Although similar approaches were used in the various phases of the human health risk assessment (*e.g.,* screening methodology, nature of multi-media pathways and routes of exposure, toxicological databases, nature of human receptors, intake factors, etc.), specific assumptions pertaining to each facility were also required. Whenever appropriate, these are reported and commented.

The present assessment attempted to estimate potential health risks associated with the atmospheric emissions, from the representative landfill, of carcinogenic and systemic toxicants; nuisance problems associated with the atmospheric emissions of odorous compounds; and the potential health impact of carcinogenic and systemic toxicants present in the raw leachate that may leak into, and be transported by, the underlying aquifer to target receptors. Although it has been recognized that other ailments and more subtle ill-defined syndromes may be associated with landfill emissions (Deloraine *et al.*, 1995; Hertzman *et al.*, 1987; Marsh and Caplan, 1987), their characterization remains hypothetical within a predictive framework in view of the unknown etiological agents responsible for these effects and the underlying mechanisms of action involved. Consequently, this aspect was not examined in detail in the present assessment. Furthermore, the potential health and comfort impact of particulate released during the operational phase of the landfill were not specifically addressed, although this should constitute a priority in any future refinement envisioned for this analysis.

These aspects were investigated with two conceptually different risk assessment approaches. The first examined individual exposures and the ensuing potential health and discomfort effects for a series of human receptors living at the location corresponding to the maximum ground-level concentration (GLC) of emitted chemicals (*i.e.,* point of maximum impact). This is termed a deterministic or "point" or reasonable worst case estimate. In the case of chemicals associated with chronic adverse effects (*e.g.,* carcinogens, systemic toxicants) this deterministic analysis was extended to incorporate a probabilistic analysis in order to obtain ranges of individual exposures. The second approach also examined, probabilistically, the potential health risks and effects for a population of approximately 550,000 living in a 20 km x 20 km perimeter surrounding the facility. This component was essential in order to characterize the extent of the impact of each facility on a broader scale and to provide a more realistic basis for comparing health risks in view of the different geographical locations associated with the maximum GLC for the landfill (near the property line) and the incinerator (within one to several kilometres downwind). This assessment contains 3 major phases, namely 1) screening, 2) modelling of concentrations in different environmental media, and 3) characterization of potential health risks and effects.

In the case of the atmospheric emissions, a multi-media approach was applied in order to examine the contribution from various environmental pathways and routes of exposure to the total intake of chemicals. These included 1) direct intake by inhalation, 2) intake following ingestion of soil and garden produce contaminated by depositions and/or air transfer of airborne emissions, and 3) intake following dermal contact with contaminated soil. The choice of these pathways was dictated by the location selected which reflects a suburban area with a large population surrounding either the landfill or the incinerator, a situation frequently observed in southern Ontario. This choice, however, precluded the assessment of pathways and/or routes that would have been considered relevant under different circumstances (*i.e.*, those associated with agricultural activities and the consumption of sport fish). The potential health impact of leachate-contaminated groundwater at the receptors location was examined by assuming direct ingestion of the groundwater. This is a worst case scenario since very few people drink well water in large suburban situations. No other pathways and routes of exposure were considered for this component as it is expected that the sub-urban residents will be served by municipal treatment plants from their or nearby communities. Indirect pathways such as vapour diffusion through soil and basement infiltration were not considered in the present analysis.

This assessment attempted to examine the potential health risks attributable to typical MSW landfill (and incinerator) emissions without considering the background levels of chemicals. Background levels are generally specific for a geographical location and, consequently, may vary considerably depending on the presence of natural and anthropogenic sources. Although the intrinsic emissions from a facility may suggest absence of significant risks for the surrounding populations and ecosystems, the presence of a higher-than-normal background of some chemicals in certain environmental media may create undesirable exposure conditions. Site-specific risk assessment of candidate sites is not required by the 1998 Landfill Standard. However, if detailed assesment of a specific candidate site is undertaken, background levels of chemicals could be addressed. As a result, and because the main objective of the present analysis was to investigate the potential baseline health risks of the facility, the background aspect was intentionally not considered.

The methodology includes three major phases, 1) screening or prioritization of the emissions and selection of a critical list of chemicals based on worst case exposure conditions, 2) modelling of the emissions into plausible multi-media pathways and exposure routes, and 3) estimation of the intake of chemicals and characterization of the potential health risks and effects for a selection of human receptors. These phases are detailed in the following sections.

B-2.2 Screening of Chemicals

Current toxicological concepts suggest that not all compounds to which humans may be exposed will create a potential health risk either as a consequence of their low intrinsic toxicity (or hazard), low quantities emitted, because of environmental factors that may minimize the amounts reaching individuals, or a combination of these factors. A screening procedure was therefore applied in order to identify chemicals of potential toxicologic concern that should be considered in the detailed risk assessment. The screening was conducted based on the assumption of dose and response additivity for both non-carcinogenic systemic toxicants and carcinogens, respectively (Seed *et al.*, 1995; CEPA, 1994; US EPA, 1986).

B-2.2.1 Screening of Air Emissions

B-2.2.1.1 Volatiles Organic Compounds (VOC's)

Given the prospective nature of this study, the identity and amounts of chemicals potentially emitted from the landfill were estimated based on data compiled for existing facilities in Ontario. In this regard, a primary list of candidate chemicals has been reported in IWA (1994). This list was the result of a review of monitoring data from four different landfills (Brittania Road, Mississauga; Essex, Windsor; Keele Valley, Vaughan; and Beare Road, Scarborough) further complemented with emission data published by the US EPA (1993). The monitoring data were originally reported as concentrations of chemicals measured in the vents (*i.e.,* beneath the landfill cover). A search of the scientific literature performed at that time also indicated that only one study had reported airborne concentrations of metals associated with total suspended dust measured at 5 different sanitary landfills in Finland (Rahkonen *et al.,* 1987).

After reviewing the original data for the Brittania landfill (RWDI, 1992) (as it provided the bulk of the data used to develop the primary list), emission data from US EPA (1993), and the results reported by Rahkonen *et al.* (1987); after evaluating the criteria used by IWA (1994) for selecting these chemicals as representative of air emissions at landfills in Ontario; and after conducting an extensive literature search which revealed that no new substantive information has been published since on the emission/resuspension/concentration of particulate-bound metals from MSW landfills, the primary list reported in IWA (1994) was adopted for the present study. This list (Table B-1) comprises 63 chemicals (56 VOC's, the nitrogen oxides class, and 6 metals) that may be emitted to the atmosphere during the operational and/or post-closure phase of the landfill. Also identified in this list are indicator chemicals recommended by the Ontario Ministry of Environment for estimating the air impact of landfill emissions and for which regulatory limits have been established (OMOE, 1992). It should be noted, however, that the quantification of these indicators under recommended procedures is not sufficient to characterize the spectrum of potential health risks associated with landfill emissions.

The VOC's (including nitrogen oxides) contained in the primary list were classified based on different end-points that included carcinogenesis, general systemic effects, and odor (see Figure 1). The carcinogens were further subdivided into *chlorinated compounds* (based on similarities in both target organs and mode of action) and *others* (*i.e.,* acting on other organ systems). Systemic toxicants were screened based on their individual effects. In addition, the qualitative and quantitative potential for all the chemicals reported in Table B-1 to induce reproductive effects, foetotoxicity and teratogenicity were also examined in view of the relevance of these end-points for exposed communities.

Chemicals were screened by comparing their 24-hour ground-level concentration modelled off-site at the point of maximum impact (24h-GLC$_{max}$) and at year 20 (year of maximum emissions from the landfill) to peer-reviewed health-based criteria or guidelines for chronic effects. In the case of the chlorinated class of carcinogens, the compounds were first ranked by decreasing order of their individual Lifetime Excess Cancer Risk (LECR); only those with a LECR greater than 10^{-6} were subsequently retained for the detailed assessment. Similarly, for carcinogens included in the *others* class only those presenting with individual excess lifetime cancer risks greater than 10^{-6} were retained. Screening of systemic toxicants known to induce sub-chronic and chronic adverse effects, including those possibly associated with reproductive toxicity, teratotoxicity, and foetotoxicity was conducted by estimating their exposure ratios (ER = 24h-GLC$_{max}$/RfC, where RfC is the reference concentration for a chemical; see Section B-2.5). In this case, only those presenting with individual ER above 0.1 were retained for further analysis.

Odorous compounds were initially screened by comparing their maximum 10 minutes ground-level concentration (10m-GLC$_{max}$) modelled off-site at the point of maximum impact with known values for odor detection thresholds. In this case, the 10m-GLC$_{max}$ was derived by multiplying the modelled 1h-GLC$_{max}$ by a factor of 6. According to Ministry modellers, a factor of 6 is generally taken to represent worst case unstable atmospheric conditions.

B-2.2.1.2 Metals

In the case of particulate-bound metals, the presence of secondary exposure pathways may contribute significantly to the total intake. Consequently, a conservative (deterministic) multi-media screening procedure was also conducted in order to evaluate the total intake for a set of routes and pathways similar to those described previously (*i.e,* direct inhalation, ingestion of contaminated soil and garden produce, and dermal exposure following contact with contaminated soil). The exposure algorithm and the associated default values proposed by the California Air Pollution Control Officers Association (CAPCOA, 1993) was used for this purpose with some modifications.

The identity of particulate-bound metals possibly emitted during the operational phase of the landfill was assumed to be identical to that reported by Rahkonen *et al.* (1987). Measurements obtained in the breathing zone of landfill workers identified cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), iron (Fe), and zinc (Zn). In order to translate the levels measured by Rahkonen *et al.*, (1987) into estimates reflective of the hypothetical landfill, weighted average concentrations were estimated for each metal from the results collected during the summer (n = 6) and autumn (n = 7) monitoring periods; these concentrations were assumed to be representative of year-round conditions. They were further standardized to the mean amount of total dust measured at site E during the same period (mean = 0.66 mg/m^3; range: 0.10 - 1.6 mg/m^3; particles > 0.8 um), thus providing a measure of the concentration of metal *per* mg total suspended dust. In view of the substantial differences in the range of total dust measured at the 5 different sites, values for landfill E were selected for this computation in view of similarities in design characteristics with the hypothetical landfill (*i.e.*, serving > 50,000 inhabitants, use of compactors and daily coverage of refuse). For the purpose of the present analysis, it was assumed that similar concentrations of metals would be present on particulate emitted from our hypothetical landfill. It was also assumed that the concentration of total suspended particulate (TSP) modelled by IWA (1994) at the southbound property line for the projected EE11 Durham landfill site (location B3) would be applicable to the present scenario. This assumption is plausible because the EE11 site is technologically identical to the hypothetical landfill. Thus similar meteorological conditions (north/west-to-south/east dominant winds) would be experienced. This was confirmed by the atmospheric modelling conducted for the present analysis which revealed a dispersion pattern comparable to that reported in IWA (1994) (see Appendix B-I). The total long-term TSP average at site B3 over the 5 periods examined by IWA (1994) was estimated to be 56 ug/m^3. Combining this value with the concentrations of metal *per* mg total suspended dust calculated previously provides an estimate of the levels of metals (in ug/m^3) potentially present at the receptor location. These concentrations were used as input in the deterministic CAPCOA model in order to estimate the integrated multi-media intake of particulate-bound metals. It should be noted that using concentrations measured in the breathing zone of workers in order to estimate those that may be present off-site at receptors location introduces a significant amount of conservatism as it is expected that only the fine particulate fractions will be translocated over some distances from the landfill, thus minimizing the total amount of metals reaching the receptors.

Other scenario-based assumptions adopted for the purpose of this screening are summarized in Table B-2. These include a) the use of maximum intake rates for human receptors (see Section B-2.3.1.3.1 for a description of receptors and AppendixB-II-A for intake rates); b) residence at the southbound property line; this location corresponds to the Annual-GLC$_{max}$, 24h-GLC$_{max}$, and 10m-GLC$_{max}$ modelled for the hypothetical MSW landfill (see Section B-2.3.1.2); c) an uninterrupted period of residence of 30 years which includes 20 years associated with the operational lifespan of the landfill when emission of particulate-bound metals occur continuously as a result of waste processing activities, and 10 years following closure of the landfill. A period of 30 years corresponds to the upper 95th$_{percentile}$ of residency period for the U.S. population (Finley *et al.*, 1994). It was assumed that direct exposure by inhalation of particulate-bound metals occurred during the first 20 years whereas exposure through secondary pathways occurred during the entire 30 years period. For this purpose, similar soil concentrations were estimated for these two periods (Section B-2.3.1.2, equation 1) as a result of the conservative default values used in the CAPCOA model for the half-life of metals in soil (1.0E+08 days or approximately 27,000 years); d) an infiltration rate (Indoor/Outdoor ratio) of 1 was selected in order to maximize exposure; e) exposure from dermal contact with soil, ingestion of soil, and ingestion of garden produce occurred during 7 days/week x 8 months/year at the maximum rates reported for the various receptors; the attenuation of 8/12 was introduced in order to take into account the winter period where contact with covered and/or frozen soil and absence of gardening activities are not expected to contribute to the total intake; and f) chemical-specific default values suggested by CAPCOA (1993) were used to estimate the amount of chemical absorbed across the skin whereas the fraction desorbed from soil particles and the fraction desorbed from plants were assumed to be 1 in all cases. Similarly, no provision was introduced to estimate the dynamics of inhaled particles and the differential deposition patterns that occur in the respiratory tract as a function of particle size.

Potential average daily doses (ADD_{pot}) of metals averaged over an exposure period of 20 years (inhalation route) or 30 years (secondary pathways) were computed for a composite receptor covering the different life stages (see Section B-2.3.1.3.1) in order to estimate exposure ratios (ER) for non-carcinogenic systemic toxicants. Potential lifetime average daily dose ($LADD_{pot}$) amortized over a period of 70 years were used to estimate lifetime excess cancer risks (LECR) in the case of metals known to induce neoplastic and/or tumorigenic effects.

B-2.2.2 Screening of Leachate Chemicals

In order for leachate components to induce adverse health effects, they have to migrate in significant amounts through several layers of protective membranes (liners) and reach the groundwater resource aquifer. Hydrogeological considerations will then dictate whether they will reach the critical receptors. Within the present analytical framework, the choice of the geographical location of the facility (sub-urban area with access to distribution networks of municipally-treated potable water) and the expected demographic expansion indicate that direct use of the groundwater for consumption purposes is unlikely. Furthermore, technological characteristics of the landfill such as the leachate collection system, the projected on-site leachate treatment facility, and the high impermeability of both the polyethylene and clay membranes suggests that only minute amounts of these components present a potential for migrating and impacting upon the surrounding environmental media (see Report A, section A-4.3). Nevertheless, the potential toxicological impact associated with direct ingestion of leachate-contaminated groundwater was examined in the present analysis.

Based on these considerations, it was believed that results already available from existing or projected state-of-the-art MSW landfills in Ontario would provide sufficient information to conduct a screening and identify compounds of concern. For reasons similar to those previously discussed in Section B-2.1, the results reported in IWA (1994) and pertaining to the projected EE11 site were used for this analysis. IWA (1994) has reported a primary list of leachate components developed based on a survey of 24 landfills located in southern Ontario (Rowe, 1994; Fenco MacLaren, 1994), including those used to develop the primary list of potential air emissions. A total of 168 organic compounds, 40 inorganic compounds of which 34 metals, and 32 leachate parameters were identified. A primary screening was conducted by IWA (1994) based on two sets of considerations. The first set incorporated conditions pertaining to a) mobility (or the potential for chemicals to migrate to the site boundary), b) frequency of detection (the probability for chemicals to be present in the leachate), c) toxic potency (the potential for chemicals to cause adverse effects). The second set of conditions involved computing and comparing the cumulative contribution of each chemical obtained from the first set (n = 57) to their combined toxic potential. A total of 38 organics and 1 leachate parameter (nitrate) were finally obtained as a result of this primary screening. These are reported in Table B-3. The long-term modelling of these 39 chemicals (including transfer across protective landfill membranes and transport in the groundwater to the receptors located at the southbound property line) was then modelled using site-specific hydrogeological conditions with a finite layer transport model (POLLUTE). Two different sets of input parameters associated with a) concentrations of chemicals in the leachate, b) soil-water partition coefficient, and c) half-life were used in order to estimate the maximum predicted concentration under anticipated conditions and under reasonable worst case conditions. A simplifying, although conservative assumption adopted by IWA (1994) for this exercise was that maximum concentrations of all leachate components would reach simultaneously the receptor location. In reality, the different rates at which chemicals move through soil and groundwater as a result of their individual physical and chemical properties would result in maximum concentrations that will unlikely coincide in time. Results of this modelling are summarized in Table B-3.

For the purpose of the present analysis, results reported for the reasonable worst case conditions were used to conduct a final screening, first by grouping chemicals according to their anticipated critical end-point (see Figure B-1), and then using pre-established cut-off values in order to retain systemic toxicants with ER greater than 0.1 and carcinogens presenting with LECR greater than 10^{-6}. These calculations were conducted for an adult receptor by assuming an average ingestion rate of 1.5 L/day (tapwater + tapwater-based beverages) and a body weight of 70 Kg (CEPA, 1994).

B-2.3 Detailed Exposure Estimates

Chemicals selected during the screening phase were used to estimate potential health risks with a probabilistic multi-media exposure model. In the case of atmospheric emissions resulting from the diffusion of gaseous constituents through the landfill cover and/or emitted from the flare as a result of incomplete combustion, the methodology included the following steps: a) modelling their temporal production in the landfill waste and their transport to and release through the landfill cover; this aspect is necessary given that the rate of production of chemicals vary significantly as a function of the amount and type of waste disposed of as well as a result of the stage of the digestion process (*i.e.*, aerobic and anaerobic phases) with peak emissions occurring approximately at time of closure *i.e.*, 20 years, b) modelling their atmospheric dispersion in order to estimate 1h, 24h, and annual average GLC at various locations around the facility, and c) applying a probabilistic exposure model to GLC at pre-selected locations (*e.g.*, point of maximum impact) in order to evaluate concentrations of chemicals in the various media of interest and estimate the combined intake of these chemicals for a series of human receptors.

In the case of particulate-bound metals, a probabilistic exposure model adapted from the CAPCOA (1993) algorithm was adapted to the scenario previously described (see Section B-2.2.1.2), whereas in the case of the leachate components selected during the screening, the maximum concentrations predicted under anticipated conditions were used without further review. Finally, the impact of odorous compounds was assessed deterministically based on the projected 10m-GLC_{max}. These approaches are summarized in Figure B-2 and a more detailed description follows.

B-2.3.1 Volatile Organic Compounds (VOC's)

B-2.3.1.1 Landfill Gas Production and Atmospheric Modelling

Technical details and results pertaining to the modelling of landfill gas production and their subsequent atmospheric dispersion are reported in Appendix B-I. Briefly, landfill methane gas (LFG) generation rates were initially estimated with the first-order decay US EPA's Landfill Air Emissions Estimation Model (LAEEM). For non-methane organic compounds (NMOCs), vent concentrations of gaseous components measured in the Brittania landfill, supplemented with other data from representative landfills located in southern Ontario (RWDI, 1992) and the US (US EPA, 1993), were combined with LFG modelling results in order to estimate individual emission rates. Several assumptions were adopted for this purpose, including a) the LFG and NMOCs were assumed to be released by diffusion through the landfill surface and by the enclosed flair, b) the collection system was not expected to be in place until the end of year 4 and, therefore, no LFG and NMOCs were assumed to be collected and flared until the end of year 5 (*i.e.,* emissions occurred exclusively by diffusion through the cover), c) collection of gaseous compounds was expected to be approximately 50% before final cover of the entire landfill surface (year 20), d) the application of a final cover was expected to be completed within 1 year after the site reached its capacity at the end of year 20, e) collection efficiency was expected to reach 70% following the completion of the final cover, and f) the enclosed flare was assumed to have a control efficiency of 98% for each individual NMOC.

Modelling was initially conducted for a period of 40 years, including 20 years of emissions corresponding to the operational phase of the landfill and 20 years of emissions following its closure. The intervals examined were 0-4, 5-8, 9-12, 13-16, 17-20, 21-25, 26-30, and 31-40 years for which the amounts estimated at the end of each period were assumed to be representative for that entire period. Emission estimates covering the first 30 years (corresponding to the assumed residency period of the receptors) were then used as input into the Fugitive Dust Model (FDM) in order to estimate 1h, 24h, and annual average GLC at pre-selected locations (approximately 600 points in 100 grid cells) within an area of 20 km x 20 km and for each time interval (*i.e.,* 0-4, 5-8, 9-12, 13-16, 17-20, 21-25, and 26-30). The possible atmospheric decay of chemicals by photooxidation, reductive reactions, and other physical and chemical processes were not considered in the present analysis. Because the estimated dispersion of chemicals was linearly related to emission rates (*i.e.,* independent of both physical and chemical properties), detailed modelling was not required for each chemical. Instead, dilution factors reflecting the change in concentration from the vent to specific locations impacted by the emissions at different time periods were obtained initially for one chemical (*e.g.,* point of maximum impact at the southbound property line). These factors were then combined with the vent concentrations of other chemicals in order to estimate their respective 1h, 24h, and annual GLC.

B-2.3.1.2 Modelling of Concentrations in Environmental Media

Volatile organic compounds (VOC's) may adsorb to suspended particulate once emitted to the atmosphere. This may influence their qualitative and quantitative transfer across environmental media and, once inhaled, the quantities deposited in various regions of the respiratory tract. Gas/particulate ratios were therefore estimated using two conceptually different methods (Mackay, 1991; Travis and Hattemer-Frey, 1989). Details pertaining to these methodologies as applied to VOC's contained in the final list of air emissions are presented in Appendix B-III-A. Results obtained indicate that only a negligible fraction of these compounds would be expected to adsorb onto airborne suspended particulate (< 0.1%). Consequently, the exposure algorithms presented in the present and the following sections assumed that the totality of VOC's were present in the gas phase.

The basic multi-media exposure model proposed by CAPCOA (1993) was adapted in order to estimate the deposition of VOC's on soil, their temporal decay, their possible uptake by root crops (*e.g.,* potatoes), and their air-to-leaf or gas-phase transfer to leafy and vine crops (*e.g.,* fruits and vegetables).

This was performed for each period (*i.e.*, 0-4, 5-8, 9-12, 13-16, 17-20, 21-25, and 26-30 years) by using the corresponding Annual-GLC_{max} at the point of maximum impact (southbound property line). For the purpose of the present analysis, Annual-GLC_{max} were assumed to be lognormally distributed with a standard deviation of 3.5, based on expert judgement by Ministry modellers. Several model parameters that differed quantitatively from the default values used in the basic CAPCOA model were used for the conduct of this probabilistic assessment. These parameters are briefly summarized in Table B-4 and reported in detail in Appendix B-II-B (statistical distributions associated with general model parameters as obtained from the peer-reviewed scientific literature) and Appendix B-III (distributions associated with specific physico-chemical properties for the selected chemicals).

Three major modifications were incorporated into the basic model to account for a) the effect of temporal emissions of chemicals on soil concentrations, b) loss through evaporative processes following deposition on the aerial parts of garden produce, and c) the assumed absence of root uptake of volatiles from leafy, vines, and root crops.

In the first case, steady-state soil concentrations were evaluated with the standard equation proposed by CAPCOA (1993) for each time period based on the Annual-GLC_{max} modelled at the end of that period. Loss of deposited chemicals during the subsequent periods was assumed to obey first-order kinetics (Borgert *et al., 1995)* according to the relation

$$A = A_0 e^{-kt} \qquad (1)$$

where A is the residual amount at time t, A_0 is the amount at steady-state during a specific period, and k is the soil half-life for the chemical (reported in Appendix B-III). Therefore, the total concentration in soil for each time period was the sum of the steady-state concentration resulting from the deposition occurring during that period plus the residual amounts resulting from the previous depositions. Hence, in the hypothetical example presented in Figure B-3, the total soil concentration estimated for the 13-16 years period is the result of 4 components, one resulting from the steady-state concentrations for the 13-16 years period plus 3 residuals (0-4, 5-8, and 9-12). A matrix describing the components used to calculate soil concentrations for each period is presented in Figure B-4. For the purpose of these calculations, residual amounts (A) were estimated at mid-period. It was also assumed that no interactions were present between chemicals being deposited and those already present on the soil surface (*e.g.*, no perturbation of the dynamic equilibrium). Furthermore, no provision was introduced to account for possible seasonal conditions that may decrease the amount of chemicals that deposit on soil (*e.g.*, snow cover and frozen soil) as it is plausible that dissolved chemicals in the snow cover, if any, could migrate/interact with surface soil at springtime. Similarly, no provision was introduced to account for the possible decrease in the loss rate of volatiles from frozen and/or covered soil.

Accumulation and loss of volatiles following their deposition and/or contact with the aerial parts of leafy and vine crops was initially estimated based on the relationship proposed by CAPCOA (1993). However, in addition to weathering and senescence, removal of volatiles from vegetation also occurs by surface vaporization (McKone and Daniels, 1991; Environ, 1986). Therefore, in order to estimate a total rate constant that would reflect these various phenomenon, a rate of vaporization was estimated for each contaminant using the fugacity-based relationship (Rv) proposed by McKone and Daniels (1991), the details of which are reported in Appendix B-III. In line with other observations (Environ, 1986), results obtained indicated that removal of volatiles by weathering and senescence is negligible when compared to evaporative loss which, thus, becomes the limiting factor for the accumulation of volatiles in plants. Consequently, only the contribution from air-to-leaf transfer, estimated by using Rv as the rate loss constant (k) in the CAPCOA relationship, was used in order to calculate the amounts of volatiles present in leafy and vine crops at steady-state.

Although experimental evidence suggests that volatiles have the capacity to partition between soil and plant roots (Hattemer-Frey *et al.*, 1990; Environ, 1986), the third modification assumed that root transfer would not occur to any significant extend. This assumption is based on the high volatility of these chemicals and, hence, their tendency to remain in the vapour phase (*i.e.*, no significant adsorption on soil particles as previously determined), and their relatively low water solubility. The combination of these factors is likely to prevent volatiles from migrating downward and reach the root zone (10-15 cm) where uptake by plants would occur. This assumption is supported by modelling results conducted for a series of volatiles (including some of those selected here during the screening phase[1]) where it was demonstrated that volatilization was the main factor responsible for loss in soil ($> 99\%$) when compared to the contributions from soil leaching and biodegradation. Based on these considerations, the root uptake pathway was not considered when estimating concentrations of volatiles present in garden produce.

B-2.3.1.3 Modelling of Individual Exposure to Volatiles and Determination of Risk Descriptors

B-2.3.1.3.1 Compounds Exhibiting Tumorigenic and/or Carcinogenic Properties

A series of human receptors representing various age groups (0-6 months, 7 months-4 years, 5-12, 13-20, and 21+) were assumed to reside during a period of 30 years at the location impacted by the 1h-GLC_{max}, 24h-GLC_{max}, and Annual-GLC_{max} (*i.e.*, southbound property line). A value of 30 years is considered conservative as it represents approximately the upper $95th_{percentile}$ of residency period estimated for the U.S. population as reported by Finley *et al.* (1994). Given that exposure was evaluated based on the potential dose (or intake) of chemicals, only female subgroups were investigated in view of their higher intake rates *per* body weight when compared to the male subgroups (see Appendix B-II-C). Statistical distributions associated with the exposure variables and used in the probabilistic assessment (see Appendix B-II-A) were those developed by IWA (1994) based on survey data from EHD (1993), OMAF/OMOE (1988), US EPA (1985), and Nutrition Canada (1975).

In accordance with actual concepts regarding the stochastic nature of the mechanisms of action responsible for the induction of carcinogenic and tumorigenic effects in mammals, the cumulative dose (in this case potential dose or intake) standardized over a lifetime period of 70 years was used as a risk descriptor. This choice was also dictated by the nature of the peer-reviewed health-based values used in the present assessment (*i.e.*, Risk Specific Doses (RsD) and Risk Specific Concentrations (RsC) developed from lifetime bioassays or from long-term retrospective studies of human cohorts; see Section B-2.5). In order to estimate potential Lifetime Average Daily Doses ($LADD_{pot}$) (US EPA, 1992) for volatiles exhibiting tumorigenic and carcinogenic properties, a composite receptor incorporating the various life stages (0-6m, 7m-4y, 5y-12y, 13y-20y, and 21y+) was assumed to be exposed to concentrations of volatiles present in environmental media (air, soil, garden produce) resulting from the emissions associated with each time period (0-4, 5-8, 9-12, 13-16, 17-20, 21-25, and 26-30). For this purpose, the composite receptor was positioned in such a way as to maximize the intake of chemicals during the assumed 30 years of residence at the site. Preliminary calculations indicated that maximum intake over the lifespan of the receptor occurs when the position of the 0-6m subgroup coincides with the beginning of the emissions (0-4 years period). This scenario, illustrated in Figure 5, indicates that sequential intakes of chemicals from inhaled air, soil and garden produce occur as follows: 0-6m and 7m-4y: during the 0-4 years period; 5y-12y: during the 5-8 years and 9-12 years periods; 13y-20y: during the 17-20 years and the 13-16 years periods; 20y+: during the 21-25 and 26-30 years periods. As previously mentioned, this combination resulted in a maximum total intake of chemicals as contributed simultaneously by all routes of exposure, a phenomenon attributable to the higher intake rate *per* body weight associated with the early stages of life (*e.g.*, 0-6m, 7m-4y, and 5y-12y). When these subgroups were positioned over periods corresponding to higher or to the maximum emissions from the landfill (13-16 and 17-20), the resulting $LADD_{pot}$ were slightly reduced as a result of the lost contribution from the 21y+ subgroup (*i.e.*, exceeds the assumed 30 years residency period). Consequently, the scenario presented in Figure B-5 was retained as it provides a more conservative exposure estimate than those obtained with other combinations.

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[1] benzene, methylene chloride, 1,1,1-trichloroethane (as a surrogate for 1,1,2-trichloroethane), perchloroethylene, and trichloroethylene

A $LADD_{pot}$ was computed for each chemical and for each route of exposure in order to estimate the individual lifetime excess cancer risks (LECR). The route-specific $LADD_{pot}$ or the associated LECR were also combined in order to provide an estimate of the total LECR whenever judged appropriate and permitted by toxicological and pharmacokinetic considerations (*i.e.*, route-independence of critical effects in target organs and tissues). The $LADD_{pot}$ for the inhalation route was estimated based on the following relationship:

$$LADD_{pot} = (\Sigma_{r=1\text{-}5} (I_r \text{ x } days_r))/AT \quad (2)$$

where I_r is the daily intake of a chemical by subgroup r (ug/kg-day), $days_r$ is the number of days associated with this subgroup (*e.g.*, 182.5 days for the 0-6m category), and AT is the averaging period (25550 days or 70 years). Summation occurs over the 5 human receptors selected for the present study. The daily intake I_r for subgroup r was estimated according to the following relationship:

$$I_r = \text{Annual-}GLC_{max} \text{ x } [IR_r ((I/O \text{ x } T_i) + T_o)]/BW_r \quad (3)$$

where IR_r is the inhalation rate for subgroup r (m^3/day) (see Appendix B-II-A), BW is the body weight (kg) associated with the subgroup (see Appendix B-II-A), I/O is the indoor/outdoor ratio, and T_i and T_o are the daily amount of time spent indoors and outdoors, respectively. Discrimination between the conduct of active and passive activities which may influence inhalation rates were not considered, and only inhalation rates associated with an active level of physical activities were used in the present analysis. For receptors exposed during more than one time period (*e.g.*, 5y-12y), I_r was estimated accordingly by using the specific Annual-GLC_{max} for each period. An I/O ratio of 0.75 was adopted in order to reflect the concentration gradient that may exist between outdoor and indoor levels of certain gaseous chemicals (Jonhson, 1994; Berglund, 1993; Hayes, 1989). A decrease of the indoor amounts may occur as a consequence of the specific reactivity of some chemicals with surfaces, the presence of sinks (*e.g.*, carpets), and the dynamics of the air exchange rates as a function of ventilation/heating and air conditioning systems, construction materials, and seasonal fluctuations. For the purpose of the present study, the I/O value of 0.75, which is an estimate of the rate of infiltration for particulate (Hayes, 1985), was selected as the most likeliest value of a triangular distribution varying from 0.2 to 1.0 (Hawkins, 1991). It should be noted, however, that these considerations apply strictly to the indoor infiltration of outdoor volatiles and do not take into considerations the contribution of indoor sources which, in some cases, may constitute the major contributors to indoor levels. The proportion of time spent indoor (0.8) and outdoor (0.2) are approximations based on a recent study of activity patterns for a series of human subgroups (Manca *et al.*, 1995). These various parameters are summarized in Table B-4.

The $LADD_{pot}$ associated with dermal contact with soils was estimated based on the relationship

$$LADD_{pot} = (\Sigma_{r=1\text{-}5} (I_r \times 0.67 \times days_r))/AT \qquad (4)$$

where 0.67 accounts for the assumption that exposure by this route and pathway would occur for 7 days/week x 8 months/year, thus excluding periods associated with frozen and/or covered ground. This assumption is considered conservative, especially when applied to the adult subgroup. The other terms are similar to those described previously for equation 2. The intake rate was estimated by using the relationship reported in CAPCOA (1993). In this case, statistical distributions were attributed to the variables representing the surface area (SA) and body weight (BW) for each subgroup (see Appendix B-II-A) and the soil loading factor (SL) (see Appendix B-II-B). The fraction of chemicals absorbed across the skin, ABS, was given a generic value of 0.03 based on the modelling results reported by McKone (1990) (*i.e.*, for organics with a Henry's Law constant $[K_h]$ of 0.1 and above). It should be noted that for the purpose of the present study the amount of chemical transferred from soil on human skin to the underlying tissue (*i.e.*, uptaken in the stratum corneum according to McKone, 1990) is considered to be a potential dose as there is no evidence in the model to suggest its subsequent diffusion into the deeper epidermal layers toward blood capillaries where systemic uptake would occur. Finally, it was assumed that the 0-6m subgroup had no significant exposure to soil through dermal contact during the first 3 months of life.

The $LADD_{pot}$ associated with ingestion of contaminated soil was estimated with equation (4) under the assumption that exposure would occur for 7 days/week x 8 month/year. I_r was estimated by using the relationship reported in CAPCOA (1993). In this case, statistical distributions were attributed to the variables representing the ingestion rate (IngR) and body weight (BW) for each subgroup (see Appendix B-II-A). The fraction of chemical desorbed from particles and bioavailable for uptake (ABS) and the fraction absorbed through the gastrointestinal (GI) tract were conservatively assumed to be 1, as recommended in the CAPCOA guidelines.

The $LADD_{pot}$ associated with ingestion of contaminated garden produce was estimated with equation (4) under the assumption that exposure would occur for 7 days/week x 8 month/year (fresh and frozen produce). I_r was estimated based on the relationship

$$I_r = [(0.47 \times C_l \times IngL_r \times FLO \times ABS_l \times GI_l) + (0.47 \times C_v \times IngV_r \times FLO \times ABS \times GI)]/BW_r \qquad (5)$$

where C_l and C_v are the concentrations of chemicals in leafy and vine crops (ug/kg), $IngL_r$ and $IngV_r$ are the ingestion rates (g/day) for receptor r of leafy and vines (represented by fruits and vegetables), respectively, and FLO is the fraction of garden produce from local origin. Statistical distributions for $IngL_r$ and $IngV_r$ are reported in Appendix B-II-A whereas a quantitative description of FLO, ABS, and GI is presented in Table B-4. The factor 0.47 was used to account for the fraction of the total mass of ingested fruit and vegetables that consist of unprotected produce (McKone and Daniels, 1991).

B-2.3.1.3.2 Compounds Exhibiting Non-Carcinogenic Systemic Properties

Volatiles in this class, including those presenting with a potential for reproductive, developmental, and foetotoxic effects, were examined with a multi-media approach only if evidence suggested that they may translocate in significant amounts and/or equilibrate within one or several environmental media and, thus reach the receptors through different exposure pathways. As an alternative, *i.e.*, if evidence suggested that these chemicals interact with human receptors mainly through the pulmonary route, then $24h\text{-}GLC_{max}$

and/or Annual-GLC_{max} modelled at receptors location during the various time periods were compared with appropriate health-based criteria or guidelines in order to estimate exposure ratios. A probabilistic approach using the distribution of GLC estimated for the various periods was used for this purpose. Whenever toxicological and/or epidemiological evidence suggested that critical effects could be associated with shorter exposure durations (*e.g.*, acute effects, dose-rate effects that favours the development or promotion of chronic effects), a different averaging time than the Annual-GLC_{max} was used. In this regard, compounds examined for possible repro/terato/foetotoxic effects were assessed based on both 24h-GLC_{max} and Annual-GLC_{max}. The former is generally considered to be a relevant metric for examining the possible effects of intense exposures during the gestational period (US EPA, 1986).

B-2.3.1.3.3 Compounds Associated with Nuisance Effects

For volatiles presenting with a potential to create undesirable nuisance effects, in this case odors, an initial screening was conducted by comparing their worst-case 10m-GLC_{max} at the receptor location with published or otherwise available geometric mean or 50% odor detection thresholds (OMOE, 1989; ACGIG, 1991). Chemicals for which exposure ratios (ER) were greater than 0.1 were retained for further assessment. In this case, a new 10m-GLC_{max} modelled at receptors location for each period (0-4, 5-8, etc.) was compared to odor detection thresholds. Values associated with the 10m-GLC_{max} were obtained by multiplying the 1h-GLC_{max} by a factor of 2, based on recommendations of Ministry modellers. A factor of 2 is generally used to reflect more realistic (stable) atmospheric conditions prevailing at the site.

B-2.3.1.4 Modelling of Population Risks

The probability that adverse effects may develop in a pre-defined number of people (or the number of possible cases) were estimated for a population living around the hypothetical landfill for a period of 30 years. Demographics associated with the community surrounding the Brittania landfill, located in the municipality of Mississauga, Ontario, were selected for this purpose. This landfill has presently a population of approximately 550,000 living within 10 km of the site. Since the population distribution around a site can vary and the patterns of concentrations for a landfill and an incinerator are different, the most appropriate estimation of the relative population risk associated with the two facilities requires that the population distribution be treated statistically. The approach used to calculate a population risk, along with an uncertainty estimate which depends on where the population is located relative to the facility included the following:

a) use of census track data for the area within 10 km of the Brittania landfill to derive a distribution of population densities averaged over 2 km x 2 km grid cells. Given the very large population around this landfill, this provides a basis for estimating a likely maximum population risk. Figure 6 shows the bimodal distribution of population density derived from the census tract data. The low density modes correspond mainly to industrial and rural areas while the high density modes represent various types of residential development;

b) the above population density data was inputted to create a customized distribution curve using the Crystal Ball software (see Section B-2.3.4);

c) air concentrations for one compound (1,1-dichloroethane)(chosen for modelling purposes) were derived as 2 km square averages over a 20 km x 20 km area around the landfill. These concentrations were then averaged over the 30 years exposure period. Since in this study the pattern of concentrations are the same for all the compounds, the estimates obtained for 1,1-

dichloroethane can be used to derive the information necessary to calculate the population risks for the other selected compounds;

c) using a Monte-Carlo sampling procedure (see Section B-2.3.4), a population density was selected from the distribution curve for each of the 100 grid cells for which concentration data is available. The selected population in each cell was multiplied by that cell concentration and the results were then summed over the 20 km x 20 km domain. The final result was a value for the population weighted exposure to a particular compound (*i.e.*, units are Population x ug/m^3);

d) the above calculation was then repeated for many random selections (100 were used) of population densities around the landfill resulting in a series of possible population exposures to concentrations. Figure B-7 illustrates the result of this analysis for population exposure to 1,1-dichloroethane. The data appear to be normally distributed;

e) Using the population concentration exposure (People x ug/m^3) derived in d) above, and multiplying the individual lifetime excess cancer risk *per* ug/m^3 (LECR x (ug/m^3)$^{-1}$), as determined with the algorithms described in the previous sections, gives the potential number of people affected following exposure during a period of 30 years to the landfill emissions. Since both these parameters are described by a distribution, the final result is a distribution of the number of people affected. This distribution would represent the combination of all the uncertainty and variability associated with the exposure component of this assessment;

f) after deriving an estimate of population risk for 1,1-dichloroethane, population risks were readily obtained for the other individual volatiles and for the whole mixture of chlorinated organics since the modelling procedures and the estimation of LECR vary linearly with concentration (in the latter case this is valid in the low dose/concentration range) and the pattern of concentrations is similar for all compounds.

B-2.3.2 Particulate-Bound Metals

A probabilistic exposure assessment was conducted in order to estimate the potential health risks associated with particulate-bound metals selected during the screening phase. For this purpose, the basic CAPCOA (1993) model and the scenario-based assumptions used for the screening phase (see Section B-2.2.1.2 and Table B-2) were modified with several of the exposure variables and model parameters previously discussed for the assessment of volatiles (see Section B-2.3.1.2 and B-2.3.1.3.1 and Table B-4). A notable exception consisted in maintaining conservative values for the soil half-life of metals in view of the scarcity of information available in the open scientific literature.

B-2.3.3 Leachate Components

Exposure to leachate constituents potentially present in the underlying aquifer at receptors location was estimated by assuming direct ingestion of contaminated groundwater. Only vinyl chloride selected during the final screening conducted in the present study was retained for this exercise. Because of the technical constraints discussed in IWA (1994), a probabilistic estimate could not be obtained for this component and, therefore, only single values suggesting potential individual health risks are presented.

In the case of compounds exhibiting tumorigenic and/or carcinogenic properties, exposure to contaminated groundwater was estimated by computing a LADD$_{pot}$ for a composite receptor as previously described. The rates of water intake reported in Appendix B-II-A (IWA, 1994) were used for this purpose. Generally, mean and maximum intake rates (L/day) were selected when data were described by a log-normal and uniform distribution, respectively.

B-2.3.4 Probabilistic Modelling

The probabilistic analysis was conducted only for the exposure component of this assessment whereas point values (*i.e.*, RfC, RfD, RsC, RsD or their equivalent) were used in the case of health-based guidelines. Probabilistic exposure estimates and potential health risks were computed with a commercial software (Crystal Ball Version 3.0, Decisioneering Inc., Denver, CO) running on an Excel spreadsheet (Microsoft Excel Version 5.0). A total of 10,000 iterations were conducted using a Monte-Carlo sampling procedure in order to solve over 160 assumptions and approximately 40 forecast cells. Parameters known to be correlated (*e.g.*, body weights *vs* inhalation rates) were arbitrarily allocated an r value of 0.67.

B-2.5 Toxicological Data

Only peer-reviewed health-based guidelines proposed by recognized regulatory agencies were used in the present analysis. Standards and other regulatory values that may include risk management considerations were not retained. A comprehensive data collection effort was conducted in order to obtain up-to-date and functional values that could be used within the multi-media framework. This implied examining information regarding route-specific toxicity and differential toxicity as a function of intensity of exposure. Whenever required and permitted by toxicological considerations, reference doses developed for one route (*e.g.*, inhalation) were used for other routes (*e.g.*, ingestion) following the application of appropriate conversion factors. For the screening phase, conversion factors simply consisted in assuming an inhalation rate of 23 m^3/day and a body weight of 70 kg. In the detailed assessment where ADD_{pot} and $LADD_{pot}$ derived from the multi-media assessment were based on units of ug/kg-day, reference concentrations (RfC) based on units of *e.g.*, ug/m^3 were converted using a composite receptor (see Section B-2.3.1.3.1). In general, receptor-based RfD/RsD's to conventional RfD/RsD's ratios were around unity and differed at most by a factor of 2. In the case of risk-based guidelines (*e.g.*, for carcinogens), a nominal level of risk of 1 in one million (10^{-6}) was adopted.

When confronted with uncertainties regarding the applicability of the available guidelines, no *ab initio* work was undertaken in order to complement the information available with more recent scientific findings. However, for the sake of conservatism, and in order to be consistent with actual public health practice, the most conservative values among those available were selected whenever judged appropriate. This was the case for compounds identified as possible foetotoxic, teratogenic, or reproductive effects (Goldberg *et al.*, 1995a) which were screened based on the most sensitive reported health-based guidelines.

Priority for the collection of health-based values was given to the following databases: a) the on-line Integrated Risk Information System (IRIS) of the US EPA, b) the Priority Substance List (PSL) developed by Health Canada within the Canadian Environmental Protection Act program, and c) the toxicological profiles developed by the International Programme on Chemical Safety, World Health Organization. When no information was available from these agencies, a second search was conducted to examine health-based guidelines proposed by progressive US state jurisdictions such as California (CAPCOA, 1993, CalEPA, 1996), New York (NYSDEC, 1991), and Massachusett (MDEC, 1990). The NATICH tables were consulted as a last resort given the regulatory nature of the reported values (*i.e.*, including risk management considerations). An account of these values as applied to the various phases of this assessment are reported in Appendix B-III-B (carcinogenic VOC's), Appendix B-III-C (non-carcinogenic systemic VOC's, including odorous compounds), Appendix B-III-D (particulate-bound metals), and Appendix B-III-E (leachate components).

B-3. RESULTS

Results of the landfill gas production and atmospheric modelling are described in Appendix B-I. Early in the study 1,1-dichloroethane was selected for air modelling purposes. Annual average concentration patterns for 1,1-dichloroethane at each of the 8 periods in the landfill's lifetime. Over the 10 km region, modelled annual average concentrations distinctly show the predominant north north west wind direction in the meteorological data set. The maximum concentrations are along the landfill property line with the overall maximum located on the southern side for the first two periods when waste is deposited in areas 1 and 2 only. As the emissions from area 3 begin to dominate in time period 4, the maximum shifts to the south west property line. Calculated air concentrations are a maximum at year 20. After this time the gas collection system is assumed to process 70% of the gas generated by the landfill (see Table B-I-2) which results in a significant drop in overall emission rates.

Maximum 24 hour average and the maximum hour average concentrations as modelled show that the wind direction frequency plays a part in the distribution of concentrations although not quite as much as for the annual average. However, the maximum hourly average concentration occurs when the worst dispersion conditions prevail (light wind speeds and stable conditions are worst case for area source emissions). These conditions can occur with any wind direction but will occur more frequently for the predominant wind directions. The highest 24 hour average concentration was about 14 times as large as the highest calculated annual average concentration. For the hourly average, the highest value was about 38 times the highest calculated annual average concentration.

B-3.1 Screening Assessment

B-3.1.1 Volatile Organic Compounds (VOC's) Exhibiting Tumorigenic and/or Carcinogenic Properties

Detailed results of the initial screening conducted on VOC's possessing tumorigenic and carcinogenic properties are reported in Appendix B-III-B. In the case of the 13 chemicals constituting the chlorinated class of compounds only the first 9 displayed individual LECR greater than the pre-set cut-off value of 1E-06. Among chemicals classified in the *others* category, only benzene was projected to exceed the pre-set screening level (LECR = 2.2E-06). Based on these considerations, the carcinogenic VOC's selected for the subsequent phase of this analysis included: 1,1-dichloroethane, trichloroethylene, 1,2-dichloroethane, 1,1,2-trichloroethane, benzene, vinyl chloride, bromodichloromethane, 1,1-dichloroethylene, 1,1,2,2-tetrachloroethane, and methylene chloride.

B-3.1.2 Volatile Organic Compounds (VOC's) Exhibiting Non-Carcinogenic Systemic and Odorous Properties

Results reported in Appendix B-III-C indicate that only methyl mercaptan, ethyl mercaptan, and hydrogen sulphide display exposure ratios (ER) greater than the pre-set cut-off value of 0.1 when examined based on their respective 24h-GLC$_{max}$. These compounds were therefore selected for the subsequent phase of this assessment. When chemicals known to exhibit some potential for the induction of reproductive, developmental, or foetotoxic effects were assembled and submitted to a similar screening (Appendix B-III-

C), only hydrogen sulphide (previously selected) displayed an ER of approximately 0.1. Consequently, none of the other compounds classified in this category were retained for further consideration. It should be noted that these chemicals were screened by using reference concentrations (RfC) associated with the most sensitive reported end-point.

Appendix B-III-C also reports the results of the screening conducted on compounds that have been associated with odor nuisance. Among the 18 chemicals initially selected based on in-house Ministry information and other databases (OMOE, 1989; ACGIH, 1991), six had ER greater than the pre-selected cut-off value of 0.1.

B-3.1.3 Particulate-Bound Metals

Using screening assumptions, the concentrations of particulate-bound metals estimated at receptor location were as follows: Fe:1.9E-01 ug/m^3: Cd: 4.3E-04 ug/m^3; Cr^{+3}: 1.5E-02 ug/m^3; Pb: 3.9E-02 ug/m^3; Cu: 7.3E-02 ug/m^3; and Zn: 8.4E-02 ug/m^3 (see Appendix B-III-D). Application of the screening-level multi-media exposure model identified Fe, Zn, and possibly Pb as elements of potential concern (ER > 1) for receptors residing at the southbound property line. These compounds were therefore examined in the detailed assessment. It is noteworthy that the major contribution to the total intake of these chemicals was associated with secondary pathways *i.e.*, ingestion of soil and garden produce, followed by inhalation of particulate, and then dermal uptake of contaminated soil. On the other hand, the LECR associated with inhalation of particulate-bound Cd was estimated to be 3E-07 whereas its combined ER associated with both ingestion through soil-based pathways (soil intake and consumption of garden produce) and dermal exposure to contaminated soil was around 0.3. Finally, the combined ER for all routes and pathways estimated for Cr^{+3} was approximately 1. In view of the conservative assumptions used in this screening, it was not deemed necessary to explore further both Cd and Cr^{+3}. Details pertaining to this modelling are reported in Appendix B-III-D.

B-3.1.4 Leachate Components

As previously indicated in Section B-2.2.2, results reported by IWA (1994) representing the reasonable worst case conditions were used to conduct a final screening for the purpose of the present analysis. Only vinyl chloride (total LECR = 2E-06) was finally selected among chemicals initially allocated to the chlorinated class of carcinogens, whereas none were selected among chemicals grouped in the *others* category. Similarly, no chemicals could be selected from the category grouping general systemic toxicants, and from chemicals displaying potential reproductive, developmental, and foetotoxic effects. These results are detailed in Appendix B-III-E.

Table B-5 summarizes the chemicals retained for the subsequent phases of this analysis.

B-3.2 Detailed Assessment - Individuals Residing at the Point of Maximum Impact

B-3.2.1 Volatile Organic Compounds (VOC's) Exhibiting Tumorigenic and/or Carcinogenic Properties

Tables 6a and 6b present a summary of the results pertaining to the Lifetime Average Daily Doses ($LADD_{pot}$) and Lifetime Excess Cancer Risks (LECR) obtained with the refined deterministic methodology, *i.e.*, when the exposure model was used based on single (mostly central-tendency) input values (see Appendix B-II). This deterministic model was used as the basis for the probabilistic exposure model by assigning statistical distributions to each of the exposure variables for the subsequent Monte-Carlo runs. Given that the exposure metric used in the present analysis is the potential dose of chemical uncorrected for bioavailability and other pharmacokinetic and metabolic considerations, the magnitude of these LECR directly reflects the amount of chemical intake ($LADD_{pot}$) by a specific route and pathway. Details on the input parameters used to calculate exposure, and other information pertaining to the concentration of chemicals in the various media are reported in Appendix B-III-F.

Based on the scenario(s) examined in the present analysis, inhalation would be expected to represent the major contributor to the total LECR for each volatile, with values ranging from 94% (1,1-dichloroethylene) up to 99% (benzene, trichloroethylene, 1,1,2,2-tetrachloroethane). The contribution to LECR associated with soil ingestion appears to be minor (< 5%) whereas intake by dermal contact with soil and ingestion of garden produce are negligible (generally less than 1%). A total LECR was estimated for each chemical under the assumption of additivity and independence of effects across routes of exposure. The results obtained indicate that all the volatiles examined are below the accepted regulatory risk level of 1E-06, with only vinyl chloride slightly exceeding this mark (1.6E-06). When the assumption of additivity was extended to the chlorinated class of carcinogenic chemicals, the total LECR obtained was approximately 4E-06 with the major contribution originating from the inhalation of chemicals (97.0%), followed by ingestion of soil (≈ 2.3%), dermal contact with soil (0.8%), and ingestion of garden produce (≈ 0.01%). A total LECR of 5.1E-08 was estimated for benzene.

Based on these results, it was deemed appropriate to examine probabilistically only the final outputs associated with the chlorinated class of chemicals. Vinyl chloride was selected in this assessment as a result of the estimated level of risk (1.6E-06), the high concentrations generally reported in landfill vents across Ontario (IWA, 1994; RWDI, 1992), its frequent detection in areas surrounding landfills (US EPA, 1993), its use as an indicator chemical to assess the general impact of landfill emissions (OMOE, 1992), and its potent genotoxic and carcinogenic properties for which it has been classified in Group 1 (carcinogenic to humans) by the International Agency for Research on Cancer (US DHHS, 1991). These results are reported in Table B-7. Mean values for LECR and for the relative contribution of the exposure routes to total LECR obtained for both the chlorinated group of compounds and for vinyl chloride were generally comparable to those previously reported in Table B-6. For total chlorinated organics (including vinyl chloride), a mean total LECR of approximately 3.6E-06 was obtained, whereas LECR associated with the high-end of the distribution of individual risks for the composite receptor were approximately one order of magnitude higher. In this regard, the 97.5th percentile was found to be associated with a probability of 1.0E-05. The LECR estimated at the 100th percentile level (*i.e.*, covering all the possibilities under the actual assumptions and scenarios) corresponds to 8.8E-05 (Figure B-8). As shown in Figure B-8, however, the frequency associated with this outcome is considered negligible as a result of the highly skewed log-normal distribution associated with total LECR. In the case of vinyl chloride, the individual LECR estimated at the 95th and 97.5th percentiles (Table B-7) were below the 1E-05 risk mark (4.2E-06 and 5.7E-06, respectively), suggesting a relatively low level of concern for this compound.

B-3.2.2 Volatiles Exhibiting Non-Carcinogenic Systemic Properties

Exposure ratios (ER) were estimated for atmospheric concentrations of hydrogen sulphide, methyl mercaptan, and ethyl mercaptan based on GLC modelled for the various periods of emission. This approach was more appropriate than the application of a multi-media framework as a result of the high volatility of these compounds, their low tendency to partition in other environmental media, and because the critical health effect used to develop reference concentrations (RfC) is based on local and narcotic effects such as irritation and inflammation of the nasal mucosa, and effects on the central nervous system (IRIS, 1995; IWA, 1994) (see Appendix B-III-C). An annual GLC was used for this purpose as the effects associated with these compounds appear to develop following sub-chronic to chronic exposure. In the case of ethyl and methyl mercaptan, the RfC reported in IWA (1994) were used. These values were derived based on the corresponding Threshold-Limit Values Time-Weighted Averages (TLV-TWA) recommended by ACGIH after correcting with an uncertainty factor of 1000. Table B-8 reports the ER generated using the probabilistic approach obtained for the 17-20 years period which is associated with maximum rates of emission of volatiles from the landfill. Exposure ratios (ER) estimated for the other periods are summarized in Appendix B-III-G. These results indicate that the highest mean ER would occur following exposure to GLC of ethyl and methyl mercaptan (0.2 and 0.3, respectively), whereas the mean ER associated with exposure to hydrogen sulphide during the same time period is approximately 20 times lower (0.01). Exposure ratios estimated at the upper-end of the distributions suggest that no exceedance of the RfC would be expected to occur at the 95th percentile level for both ethyl and methyl mercaptan whereas an ER of 1.7 was estimated at the 97.5th level for methyl mercaptan.

B-3.2.3 Compounds Associated with Nuisance Effects

Exposure ratios were estimated deterministically for the 6 volatiles previously selected (see Section B-3.1.2) by comparing their 10m-GLC_{max} associated with each emission period at receptor location with geometric mean or 50% odor detection thresholds (reported in Appendix B-III-G). Results presented in Table B-9 for ethyl and methyl mercaptan indicate that 10m-GLC_{max} in excess of odor detection thresholds (*i.e.*, ER > 1) are likely to be experienced by receptors living at the southbound property line during each emission period associated with the lifespan of the landfill. In the case of the other chemicals, ER are at or below unity. However, the possible occurrence of odor problems can not be discarded given that ER were estimated based on geometric mean or 50% odor detection thresholds, implying that some (sensitive) individuals will still detect these chemicals when exposed at 10m-GLC_{max} associated with ER ≤ 1.

Exposure ratios (ER) were usually lower at the beginning and at the end of the 30 years period, reaching a peak (as high as 27 in the case of ethyl mercaptan) at the end of the 13-16 years period. These results suggest that odor problems associated with emissions of ethyl and methyl mercaptan are likely to be experienced at the property line and beyond. It is possible that other compounds reported in Table B-9 will generate similar problems. Finally, when ER were estimated based on 1h-GLC (which is 10m-GLC/2), the values obtained were still above unity for these two compounds, suggesting that exposure at concentrations above detection thresholds may persist for longer periods of time.

In order to examine the impact of odorous compounds on residential areas surrounding the landfill, additional modelling was conducted to determine the 10-minutes GLC (10m-GLC) for ethyl mercaptan during the 13-16 years period over the 20 km x 20 km area. These projections, shown in Figure B-9, indicate that although 10m-GLC decrease significantly as a function of increasing distance from the source (compare with the 24 ug/m^3 reported in Table B-9), ER estimated at the periphery of a 2 km x 2 km area adjacent to the source (*i.e.*, where intensive residential development would be expected) ranges between 0.2 and 0.4. These levels will most likely be detected by some residents. Exposure ratios (ER) estimated at the periphery of the immediate 4 km x 4 km area are around 0.1. If these results are scaled linearly for the other periods of emissions (based on the linearity of the modelling procedure) the range of ER at the periphery of the immediate 2 km x 2 km area would be: for the 1-4 years period: < 0.1 - 0.1; 5-8 years: < 0.1 - < 0.1; 9-12 years: 0.1 - 0.2; 17-20 years: 0.1 - 0.2; 21-25 years: 0.2 - 0.3; and 26-30 years: <

0.1 - < 0.1.

B-3.2.4 Particulate-Bound Metals

Results of the probabilistic assessment for selected particulate-bound metals that may impinge on receptors residing at the southbound property line are reported in Table B-10. Potential average daily doses (ADD_{pot}) were estimated for each pathway and its associated exposure route(s) in order to evaluate their relative contribution. Details on the input parameters used to calculate exposure and other information pertaining to the concentration of chemicals in the various media are reported in Appendix B-III-H.

The application of a probabilistic methodology that included more realistic assumptions than those previously used for the screening procedure resulted in a dramatic decrease of the ER for Fe, Zn, and Pb (see Section B-3.1.3). In this regard, ER associated with the 95th and 97.5th percentile of the resulting distributions obtained for Zn and Pb are below unity, suggesting absence of potential health effects. In the case of Fe, the upper-bound ER were around 4 (95th percentile) and 5 (97.5th percentile). The mean and median values estimated for this element, however, were similar to the reference dose (RfD), *i.e.*, ER ≈ 1.

Figure B-10 and B-11 show a breakdown of the contribution of each route/pathway to the total ADD_{pot} for Fe and Pb, respectively (data for Zn were found to be comparable to those for Fe). In the case of mean values, ingestion of garden produce appears to represent the major contributor to total exposure, followed by inhalation of particulates. The contribution of contaminated soil through direct ingestion and dermal uptake is considered minor in the case of Zn (≈ 2%) whereas it appears to be more substantial in the case of Pb (≈ 6%). These trends are amplified when the same routes and pathways are examined at the 97.5th percentile level. However, these estimates must be interpreted with caution given that certain model parameters and scenario-based assumptions used for this analysis are considered highly conservative and may not reproduce entirely the dynamics of these metals in soil and vegetation (see Discussion).

B-3.2.5 Leachate Components

Results presented in Table B-11 suggest that individual LECR for the composite receptor exposed to groundwater under anticipated conditions would be insignificant (vinyl chloride risk = 4.0E-10). This corresponds to $LADD_{pot}$ of 7.7E-07 ug/kg-day, respectively, for which the major contribution occurs during the first 12 years of life.

B-3.2.6 Population Risk Estimates

Estimates of population health risks were computed based on a sample of approximately 550,000 assumed to reside in a 20 km x 20 km area surrounding the landfill. Based on the results obtained previously, population risks were estimated only for the chlorinated class of carcinogenic compounds. The methodology used to develop this concept was previously described in Section B-2.1.3.4. The basic spreadsheet used to estimate the individual risk *per* ug/m^3 of emitted chemical and mean values for each distribution of ug/m^3 x people are reported in Appendix B-III-I. Table B-12 summarizes the essential information obtained for each chlorinated volatile as well as for the whole mixture. In general, it is noted that values for population risks were well below unity in all cases and for all the domains of the distributions (*i.e.*, from 50th up to 100th). In this regard, the highest value obtained (0.6) was for the whole mixture at a level corresponding to the 100th percentile. The practical implication of these results is that there would be no cases of adverse health effects among a population of approximately 550,000 residing within a 20 km x 20 km area around the landfill. Figure B-12 illustrate the distribution representing the potential number of cases.

B-4.0 DISCUSSION

The assessment of potential health risks associated with any source emitting a mixture of substances through a variety of pathways is a complex task. Within a predictive framework, these difficulties are amplified by the need to resort to several modelling techniques which require the incorporation of numerous assumptions and default values. As a result, a substantial amount of uncertainty generally accompanies final estimates of exposure and risks (Cullen, 1995; Hattis and Burmaster, 1994). An additional, and perhaps more important element contributing to this uncertainty is associated with the precept, imbedded into the risk assessment paradigm, that health effects can be quantified using dose-response curves of single agents obtained mostly from animal studies. It is well known, however, that the spectrum of health responses associated with mixtures of agents such as those emitted from a MSW landfill may be influenced by subtle physical, chemical, pharmacokinetic and biological interactions that occur between the individual constituents and which may translate into synergistic and/or antagonistic effects (Seed *et al.*, 1995; US EPA, 1986). The uncertainties associated with the extrapolation to humans of toxicity data obtained from animal bioassays are also well known and have been extensively discussed (Calabrese, 1988; 1983). Nonetheless, despite these limitations modern risk assessment techniques and their variants remain the preferred and, perhaps, the only approaches currently available to regulators and public health officials in order to make inferences on the likelihood of occurrence of cancer or systemic dysfunctions.

The present analysis attempted to evaluate these outcomes by using a classical risk assessment framework. In the screening phase, chemicals were grouped into general classes of end-points; individual ER or LECR were computed based on 24h ground-level concentrations (or from a conservative multi-media exposure algorithm in the case of particulate-bound metals) modelled at the point of maximal impact and for the period corresponding to maximum emissions; and the screening was conducted using the assumption of dose and response additivity based on acceptable cut-off levels for both exposure (ER = 0.1) and risks (LECR = 1E-06). In the second phase, two conceptually different assessments were conducted. The first examined individual health risks for a series of human receptors living at the property line whereas the second attempted to make projections for an entire population assumed to live in a 20 km x 20 km perimeter surrounding the hypothetical landfill. In both cases, a probabilistic Monte-Carlo sampling procedure was used to generate distributions of potential doses that served as input for the calculation of health risks. The advantages offered by probabilistic modelling techniques have been extensively promoted as a replacement for the traditional point-based estimates in view of their accrued realism (Whitmyre *et al.*, 1992a,b; US EPA, 1992; NRC, 1991). Nevertheless, several conservative assumptions were used in the present analysis. These included aspects regarding the behaviour of people, their exposure conditions, cancer potency values, and certain parameters used to model the movement of chemicals in environmental media. These assumptions are summarized in Table B-13.

In some cases, the information necessary to characterize these parameters was not readily available (*e.g.*, amount of chemical desorbed from particles in the GI tract, half-life of metals in soil, distributions associated with cancer potency factors) and, consequently, conservative assumptions were adopted. For other parameters, conservative assumptions were deliberately incorporated despite the availability of statistical distributions (*e.g.*, time of residency at one location) in order to facilitate the calculation of exposure for the composite receptor. It was envisioned, however, that additional time and resources would be invested to investigate further the statistical nature and the impact of these parameters on the final results in the event that risk projections were judged unacceptable.

Despite these limiting assumptions, the results obtained suggested that chemical emissions from the hypothetical landfill would not be expected to impact adversely upon individuals and populations living around the facility. For the majority of scenarios examined, the projected health risks were well within the limits generally considered acceptable by regulatory agencies. Hence, the distribution of lifetime excess cancer risks (LECR) estimated for individuals living at the property line and assumed to be constantly impacted by the mixture of volatile chlorinated organics during a period of 30 years varied between 4E-06 (mean) and 1E-05 (97.5th percentile). In the case of non-carcinogenic systemic toxicants classified as organic volatiles and particulate-bound metals, mean, 50th, and upper-bound ER were generally around or below unity. Similarly, individual LECR associated with the ingestion of groundwater contaminated by leachate components under anticipated conditions were only about 1E-10 (vinyl chloride). Finally, estimates of potential health risks for a population of approximately 550,000 residing within a perimeter of 20 km x 20 km around the landfill and assumed to be impacted constantly during a period of 30 years suggest that there would be no cases of adverse effects of the nature of those associated with the individual chlorinated VOC's and the mixture as a whole under the assumption of response additivity (US EPA, 1986).

Some exceedances were noted for Fe ($ER_{95th} = 4$; $ER_{97.5th} = 5$) and for compounds associated with odor nuisance. The significance of these projections, however, must be interpreted in light of various considerations.

In the case of Fe, exceedances in the range of 4 and 5 over the reference mark are considered small in view of the conservatism associated with the assumptions used in the exposure algorithm and the magnitude of the safety and uncertainty factors generally incorporated in these exposure limits. Furthermore, these ER correspond to upper-bound values obtained from a highly skewed log-normal distribution, thus suggesting that their frequency of occurrence is very small. Two major factors, however, may help explain these exceedances. The first is associated with the fact that the annual concentrations of metals estimated off-site at the point of maximum impact were derived from results of personal monitoring of total suspended dust (particles $> 0.8\ \mu m$) obtained in the breathing zone of landfill workers. In addition, the computation of these values was based on monitoring results obtained during summer and autumn, *i.e.*, at a time when the ground is relatively dry and uncovered, thus favouring the resuspension of coarse particles by mechanical activities. Because of these factors, it is believed that the concentration of particulate-bound metals estimated off-site at receptors locations might have been greatly overestimated as it is expected that 1) only the finer fractions of TSP will translocate over some distances from the source[2] whereas coarse particles would deposit on-site, and 2) lower amounts of total dust would be generated during winter and spring thus contributing to lower overall annual levels off-site. It should also be noted that the use of TSP-based estimates to calculate the amount of particulate-bound metals intaken by receptors introduces an additional degree of conservatism given that the respirable fraction (*i.e.*, particles $\leq 10\ \mu m$ or PM_{10}) is the major determinant governing the amount of particles entering and depositing in the respiratory tract. In this regard, it has been reported that the fraction of mean PM_{10} to TSP (PM_{10}/TSP ratio) varies from 0.8 to 0.2 for TSP values ranging from $< 50\ ug/m^3$ to approximately

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[2] considering that there is a minimal buffer zone of 100 meters between the area associated with landfilling activities and the property line

400 ug/m^3 (Dann, 1994). The mean on-site concentration of total dust measured by Rahkonen *et al.* (1987) and used for the present exercise (660 ug/m^3; see Section B-2.2.1.2) suggests that in this case the PM_{10} fraction could have been more than 5 times lower the nominal TSP concentration.

The second factor that can be invoked to rationalize the exceedances observed for Fe is the conservatism associated with the use of the annual air guideline for iron compounds (1 ug/m^3) derived by the state of Texas (NATICH, 1992) and used in the present analysis to estimate an oral reference dose (RfD) (other annual air guidelines available in the NATICH database were those for iron oxide fumes: 5 ug/m^3 and iron pentacarbonyl: 0.8 ug/m^3). Because an RfD for the oral route was not available in the major databases consulted, a composite receptor-based RfD (see Appendix B-III-H) was estimated from the air guideline, thus resulting in a value of 0.42 ug/kg-day. This exposure limit, which was used to estimate the ER for ingestion of contaminated soil and garden produce in the multi-media analysis, is considered highly conservative given that Fe is an essential element for which the minimum daily requirement ranges from about 10 to 50 mg/day depending on age, sex, physiological status and bioavailability (WHO, 1993). More specifically, when recommended daily dietary allowances (Ganong, 1989) were standardized to body weight for the various subgroups investigated in this analysis, the results obtained were as follow: infant 0-6m: ≈ 2.4 mg/kg-day; children 7m-4y: ≈ 1.2 mg/kg-day; children 5-12: ≈ 0.5 mg/kg-day; teenager 13-20: ≈ 0.2 mg/kg-day; and female 20+: ≈ 0.25 mg/kg-day. This indicates that the estimated RfD was between 600 and 6000 times lower than the amounts corresponding to daily nutritional requirements. Based on results reported in Table B-10, the total ADD_{pot} of 2 ug/kg-day associated with the 97.5th level (of which more than 80% are contributed by ingestion of garden produce) represents between 3.3E-04 % and 3.3E-05 % of the total daily nutritional intake of Fe for these subgroups. Based on these considerations, it is believed that the use of a more realistic RfD for ingested Fe, once appropriately standardized for daily nutritional requirements, would result in a significant reduction of the ER.

As a corollary, consideration must also be given to the applicability to the Ontario context of the data reported by Rahkonen *et al.* (1987). A study conducted approximately during the same period by the Ontario Ministry of Labour (Mozzon *et al.*, 1987) has examined the elemental content of bulk samples of cover soil from 3 landfills in Ontario. A personal air monitoring program for on-site workers was also conducted for the presence of barium (Ba), lead (Pb), chromium (Cr), cadmium (Cd), selenium (Se), tellurium (Te), and antimony (Sb). The results obtained indicated that these elements were below detection limits in samples of cover soil whereas airborne Pb ranged from undetected to a maximum of 3 ug/m^3. None of the other metals were detected in air samples, thus suggesting that the levels of particulate-bound metals estimated at receptor location for the present analysis may not have been underestimated.

In light of the various factors discussed above, it is believed that the exceedances observed for Fe are not toxicologically relevant under the assumptions adopted for the present analysis.

In the case of odorous compounds, the results obtained indicated that significant exceedances of the odor detection thresholds must be expected for ethyl and methyl mercaptan at the site of maximum impact during the assumed 30 years of residency. Furthermore, odor problems can not be discarded for the other compounds examined as the detection thresholds used in this analysis are geometric means or 50% detection limits (OME, 1989; ACGIH, 1991). As a result, sensitive individuals may still perceive these odors when present at lower concentrations, a situation likely to occur for the other chemicals reported in Table B-9. The additional modelling conducted for ethyl mercaptan over the 20 km x 20 km area surrounding the landfill suggests that odor problems will most likely be experienced, to a lesser or greater extent, by the entire community. It should be noted that these estimates were derived based on stable meteorological conditions. Consequently, changes in these conditions will alter the pattern of atmospheric dispersion of the emitted chemicals and, hence, the location of the areas impacted as well as the resulting concentrations at ground-level. Under certain circumstances, this could result in more people being exposed during longer periods and at higher concentrations of odorous chemicals. There exist also some experimental evidence to suggest that exposure to mixtures of odorous compounds will either increase or decrease the intensity of the odor associated with each individual constituent of the mixture. Amoore (1985) has reported that under the rule of additivity of odor units ($U = C/C_t$, where U is the odor unit, C is the concentration, and C_t is the threshold), the total odor units of a mixture, represented by $U_{mixture} = U_1 + U_2 + \ldots$, may be used to predict quite accurately the intensity of binary mixtures. Laing *et al.* (1994) have tested different combinations of a quaternary odor mixture consisting of unpleasant constituents and provided evidence of infra-additive[3] effects. However, as the number of constituents in a mixture increased, the intensity of the most dominant component provided a good approximation of the intensity of the mixture. The implication of these findings for the present analyses are not clear although they suggest that the occurrence of episodes of odor intensities higher than those predicted based on individual odor thresholds can not be discarded.

The overall relevance of these predictions regarding health and nuisance outcomes associated with the hypothetical MSW landfill depends largely on the validity of the methodological framework used to conduct this analysis. Several technical assumptions such as the nature of the compounds emitted, estimates of ground-level concentrations across the impacted area, the multi-media exposure algorithm, and the nature of the selected exposure routes and pathways are not expected to distort significantly the quantitative estimates of risk in view of 1) the comprehensive nature of the methodology used to generate the candidate list of chemicals, 2) the use of validated and accepted dispersion and multi-media models, 3) careful selection of the exposure variables and parameters, and 4) the adoption of a probabilistic framework which provides a continuum of plausible exposures for the receptors examined. However, other factors may affect the validity of these conclusions in a way that may not be quantifiable. These include the omission of certain exposure routes and pathways, the use of conceptual assumptions such as dose and response additivity for addressing the potential health impact of complex mixtures, and limitations associated with the nature of the end-points that can reasonably be predicted with prospective health risk assessment methodologies. These aspects are briefly discussed below.

Sources of Uncertainty

Exposure Routes and Pathways

Given the nature of the geographical location selected for the MSW landfill, it is conceivable that the inclusion of two additional exposure pathways, *i.e.*, exposure through breastmilk in the case of the 0-6 months subgroup, and ingestion of indoor dusts originating from infiltrated particulate material may have provided a more comprehensive exposure framework. However, the influence of these components on the final risk estimates is arguable, firstly because the present analytical approach used the intake of chemicals as an exposure metric in order to estimate potential health risks (although it is recognized that this methodology does not account for differences that may exist in the disposition and interaction of chemicals in neonates and infants as compared to adults and which may result in a different (and unpredictable) set of adverse effects (Bearer, 1995; Milsap and Jusko, 1994)). Secondly, carcinogenesis was the critical end-point identified for the selected chlorinated compounds (*i.e.*, compounds most susceptible to bioaccumulate in lipids and be transferred to neonates in this case). As a result, the contribution to the total $LADD_{pot}$ during a fraction of the lifetime (0.7 % in the case of the 0-6 months subgroup) would be expected to be significantly dampened once integrated over the 70 years lifespan, thus minimizing the importance of this pathway. In the second case, ingestion of indoor dusts originating from infiltrated particulate would be expected to influence principally the total intake of particulate-bound metals in view of the negligible amounts of VOC's vehicled through this pathway. The results for Fe and Pb presented in Figure B-10 and B-11 suggest, however, that this contribution would be at best minimal (*e.g.*, when combined with the more important contribution from direct soil ingestion) and, consequently, should not alter significantly the final estimates of ADD_{pot}. Based on these arguments, it is believed that any uncertainty introduced by the omission of the two pathways discussed above should not affect significantly the overall conclusions of this analysis.

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[3] the perceived odour intensity of the mixtures was equal or greater than that of any of the individual constituents, but less than the sum of their intensities.

Assumption of Dose and Response Additivity and Nature of Adverse Effects

Assumptions of dose and response additivity introduce a substantial amount of uncertainty in any health risk assessment. Consequently, the possible impact that this premise may have on the conclusions reached in the present study is briefly examined within a weight-of-evidence approach using both experimental and epidemiological data.

Interactions between the constituents of a mixture can take place at any stage of their physiological handling (*e.g.*, absorption, distribution, metabolism, elimination, interaction with critical receptors and cross-influences) and also within the mixture itself where physical and chemical processes can affect, qualitatively and/or quantitatively, the identity of mixture components (*e.g.*, creation of new components such as breakdown products). Although some biochemical and toxicologic responses can be hypothesised with a certain degree of accuracy in the case of simple binary mixtures (Mumtaz *et al.*, 1993; Mumtaz and Durkin, 1992; Calabrese, 1991) and for mixtures of similarly acting compounds (Barnes, 1991), the nature of the adverse outcomes associated with exposure to complex environmental mixtures is beyond what can be predicted with current toxicological principles. In view of this situation, the concept of dose and response additivity for compounds acting in similar target organs through purportedly similar mechanisms has been generally accepted as a default regulatory approach by the scientific community (CEPA, 1994; CAPCOA, 1993; US EPA, 1986).

The epidemiological evidence indicates that there is widespread consensus on the fact that measured or otherwise modelled concentrations of air- and water-borne toxicants originating from these sites are well below those known to cause symptoms by recognized toxicological mechanisms (Deloraine *et al.*, 1995; Shusterman, 1992; Shusterman *et al.*, 1991; Lipscom *et al.*, 1991; Dunne *et al.*, 1990; Baker *et al.*, 1988; Hertzman *et al.*, 1987; Ozonoff *et al.*, 1987; Roht *et al.*, 1985). Self-reported symptoms identified in these communities include fatigue, headache, and nausea; skin, eye, throat, and pulmonary irritation; changes in mood; sleep disturbances, and exacerbations of underlying medical conditions. In a few instances, a higher incidence of pregnancy outcomes have also been reported although the significance of these observations was inconclusive in view of the possible influence of unaccounted factors and low statistical power (Goldberg *et al.*, 1995b; Dunne *et al.*, 1990). Very few of these studies have characterized the nature and the level of exposure experienced by these communities in a way that could be compared with the projections reported in the present analysis. A notable exception is the recent case-control study conducted by Deloraine *et al.* (1995) which used the incidence of ailments recorded by physicians in a community exposed to residual emissions of an inactive hazardous waste landfill. Although the landfill had been closed approximately 2 years before the conduct of the study, odor problems and public concern triggered the intervention of health officials. Through chromatography/mass spectrometry the authors identified approximately 300 VOC's. Using a combination of atmospheric dispersion modelling, activity patterns and questionnaires, no statistically significant increase in the reporting of ailments specified as incident cases (*i.e.*, the patient had not been cared for its condition previously) were noted for people exposed to increasing concentrations of the mixture (0-100, 101-300, and > 301 ug/m^3). Increases were noted in the reporting of ailments specified as prevalent cases (*i.e.*, the patient was known with this condition before he/she was included in the study), although none were statistically significant at the 0-100 and 201-300 ug/m^3 levels. The prevalent symptoms reported were mainly of the type of those identified in other studies, *i.e.*, irritative respiratory conditions and psychologic disorders (*e.g.*, insomnia, neurosis, anxiety).

Two observations are particularly relevant to the present discussion. The first indicates that concentrations of VOC's totalling 300 ug/m^3 were not associated with an increased reporting of prevalent and incident symptoms despite the presence of a diversity of chemicals known to induce a large spectrum of health effects when present at higher concentrations. These are reported in Table B-14 where the amounts measured by Deloraine *et al.* 100 metres from the site are compared against the 24h-GLC$_{max}$ projected at the property line (≈ 100 metres) of the hypothetical landfill at year 20. It is noted that the total amount of VOC's are 4 times higher in the case of Deloraine *et al.*, a result largely influenced by saturated aromatic hydrocarbons. Exclusion of this class would result in comparable amounts of VOC's (*i.e.*, 131 *vs* 119 ug/m^3). Of major interest, however, is the resemblance observed among measured and modelled concentrations for the other classes of chemicals for which results were available. An exception is noted for sulphur compounds which are predicted to be approximately 13 times higher around the hypothetical landfill at year 20. The second observation is the interpretation given by the authors (based on the actual and previous studies) that the reporting of prevalent symptoms observed above 300 ug VOC's/m^3 could reflect an acquired condition originating from past exposures, *i.e.*, at a time when the landfill was still active and exposure levels were higher. It was concluded that the actual levels of VOC's (up to 300 ug/m^3) would not be expected to represent a causal factor for the prevalent symptoms reported, thus suggesting the innocuity of the VOC's levels predicted for the hypothetical landfill. However, these interpretations must be tempered by the methodological limitations of the study of Deloraine *et al.* such as the possible presence of less hazardous compounds in each or some of the chemical classes investigated by these authors, insufficient statistical power and limited duration of the study, and relevance of the approach used to select cases and controls. Furthermore, recall bias can not be excluded as a possible confounding factor in view of the high emotional state surrounding this issue among community members (Neutra *et al.*, 1991; Roht *et al.*, 1985). Nonetheless, these observations are consistent with other community health studies both in the pattern of self-reported symptoms and absence of toxicological and clinical outcomes. As such they support the qualitative and quantitative predictions obtained from the toxicological analysis.

An additional element which reinforces these conclusions is presented in Figure B-13. It is noted that maximum 24-hour ground-level concentrations modelled at the property line and at year 20 (24h-GLC$_{max}$) are either within the range of background values or below methods detection limits for a series of volatiles measured in downtown metropolitan Toronto (IWA, 1994). Although these background values generally are not representative of those present in sub-urban areas (*i.e.*, at the landfill location), they indicate that landfill emissions would not be expected to impinge significantly upon the actual outdoor levels of VOC's experienced by communities. It should also be noted that annual averages modelled beyond the property line and at time-periods other than 16-20 years would be significantly lower than the 24h-GLC$_{max}$ reported in Figure B-13.

The occurrence of incommodious effects can not be discarded. Although the results reported by Deloraine *et al.* (1995) suggest that subtle physiological disturbances of the type of those associated with self-reported symptoms would be unlikely to occur following exposure to concentrations attaining 300 ug VOC's/m^3, the predicted levels of sulphurous compounds and the modelling results discussed in the previous section indicate that odor problems are likely to be experienced by the community residing around the hypothetical landfill. The role of odors as an effect modifier has been largely recognized. In fact, a common link among the majority of health studies conducted to date on communities living around landfills was the detection of odorous compounds possessing odor thresholds 3 to 4 orders of magnitude lower than their thresholds for respiratory irritation or systemic toxicity. It has been postulated that odors may serve as a sensory cue for the manifestation of autonomic or stress-related symptoms (*e.g.*, headache and nausea) among individuals concerned about the quality of their neighbourhood environment (Shusterman *et al.*, 1991). Furthermore, the observation that irritative symptoms (eye and throat) are elevated in a similar pattern with respect to this variable and that odor perception by control groups results in a similar or higher frequency of symptoms reporting was interpreted as evidence that odor and worry heighten symptoms perception or recall (*i.e.*, results in recall bias) (Shusterman *et al.*, 1991; Neutra *et al.*, 1991). Several mechanisms have been suggested to explain these observations (Shusterman, 1992) although none appear to be entirely satisfactory. Much work remains to be conducted in order to clarify the pathogenesis of odor-related effects and their relevance, if any, for communities exposed to landfill emissions.

B-5.0 CONCLUSIONS

In summary, based on these lines of evidence, it can be predicted with a reasonable degree of confidence and within the limits imposed by the methodology used in the present analysis, that emissions originating from the hypothetical MSW landfill would not be expected to result in toxicological or adverse clinical outcomes for the surrounding community. However, the perception and/or detection of odors by individuals residing in proximity of the hypothetical MSW landfill may result in a pattern of complaints and self-reported symptoms of the type of those described in the previous section.

These projections find support from the practical experience gained by public health and government officials who have witnessed a high reporting of odor complaints from communities residing in proximity of MSW landfills in Ontario. More importantly, they are confirmed by numerous community health studies published in the peer-reviewed scientific literature on hazardous and sanitary waste sites which have reported that the prevalence of overt adverse effects (*e.g.*, cancer, total mortality, and reproductive outcomes) were not higher in the exposed than in the control populations whereas there was generally a significant increase in the complaints associated with odors and subjective symptoms.

The message conveyed by this analysis must be interpreted with caution when attempting to evaluate other scenarios as the conclusions obtained were for existing or proposed state-of-the art MSW landfill (prior to the 1998 Landfill Standard) of pre-defined capacity equipped with gas and leachate collection and treatment systems which meet specific criteria of efficiency and reliability. In addition, the location selected for the landfill dictated the routes and pathways of exposure examined and the nature of the impacted population as a result of site-specific considerations such as hydrogeology, prevailing meteorological conditions, surrounding ecosystems, and projected demographics.

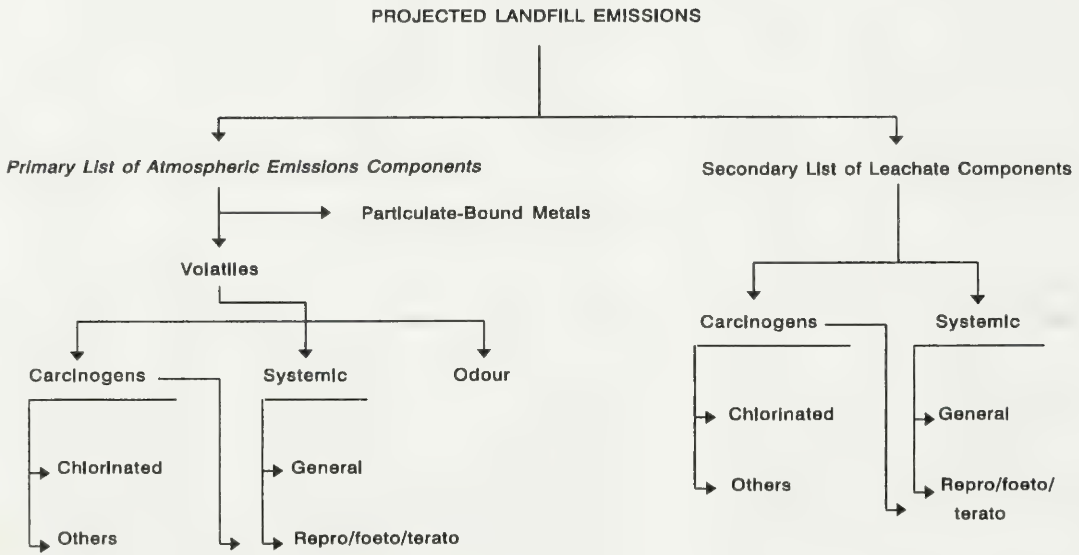


Figure 1. Screening Procedure - Classification of Chemicals

Figure B-1. Screening Procedure - Classification of Chemicals

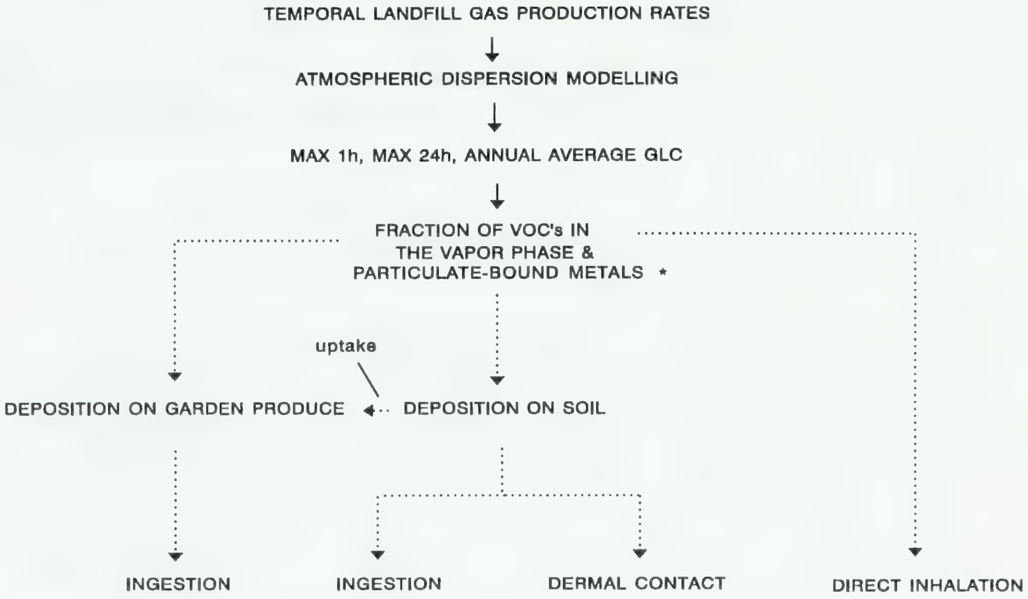

TEMPORAL LANDFILL GAS PRODUCTION RATES
ATMOSPHERIC DISPERSION MODELLING
MAX 1h, MAX 24h, ANNUAL AVERAGE GLC
FRACTION OF VOC's IN THE VAPOR PHASE & PARTICULATE-BOUND METALS *
uptake
DEPOSITION ON GARDEN PRODUCE
DEPOSITION ON SOIL
INGESTION
INGESTION
DERMAL CONTACT
DIRECT INHALATION

Figure 2. **Routes and Pathways used in the Refined Exposure Assessment**
(dotted-lined components were assessed deterministically and probabilistically)
* Off-site levels based on Rahkinoen et al. (1987)

Figure B-2. Routes and Pathways used in the Refined Exposure Assessment

Concentration in Soil at End of Period

	4	8	12	16	20	25	30	40
GLC 0-4	X	C	C	C	C	C	C	C
GLC 5-8		X	C	C	C	C	C	C
GLC 9-12			X	C	C	C	C	C
GLC 13-16				X	C	C	C	C
GLC 17-20					X	C	C	C
GLC 21-25						X	C	C
GLC 26-30							X	C
GLC 31-40								X

Figure 4. Approach Used to Estimate Soil Concentrations for Each Time Period

(X: based on the modelled GLC; C:contribution from decay)

Figure B-4. Approach used to estimate Soil Concentratations for each Time Period

for an Hypothetical Chemical

Figure B-3. Estimation of Soil Concentrations for the 13-16 Time-Interval for a Hypothetical Chemical

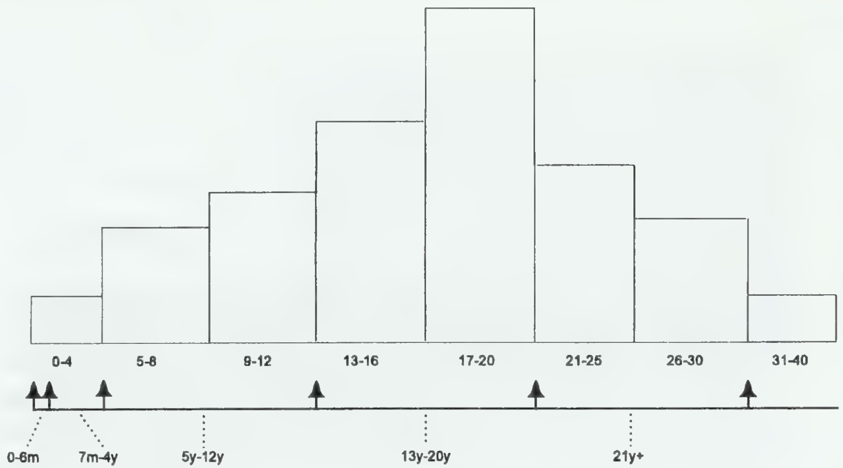


Figure 5. Positioning of Human Receptors as a Function of Temporal Air Emissions from the Hypothetical Landfill Assuming an Exposure Period of 30 years

Figure B-5. Positioning of Human Receptors as a Function of Temporal Air Emissions from the Hypothetical Landfill Assuming an Exposure Period of 30 years.

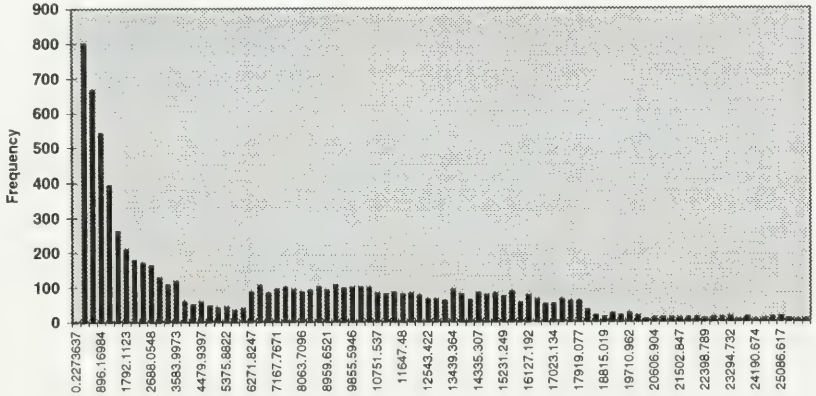


Figure 6. Distribution of Population Density Around The Brittania Landfill

Figure B-6. Distribution of Population Density around the Brittania Landfill

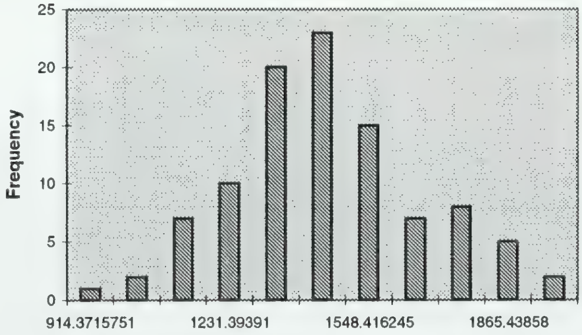


Figure 7. Distribution of ug/m3 x People for 1,1-Dichloroethane Modelled for the Area Impacted by Landfill Emissions

Figure B-7. Distribution of ug/m^3 X People for 1,1-Dichloroethane Modelled for the Area Impacted by Landfill Emissions

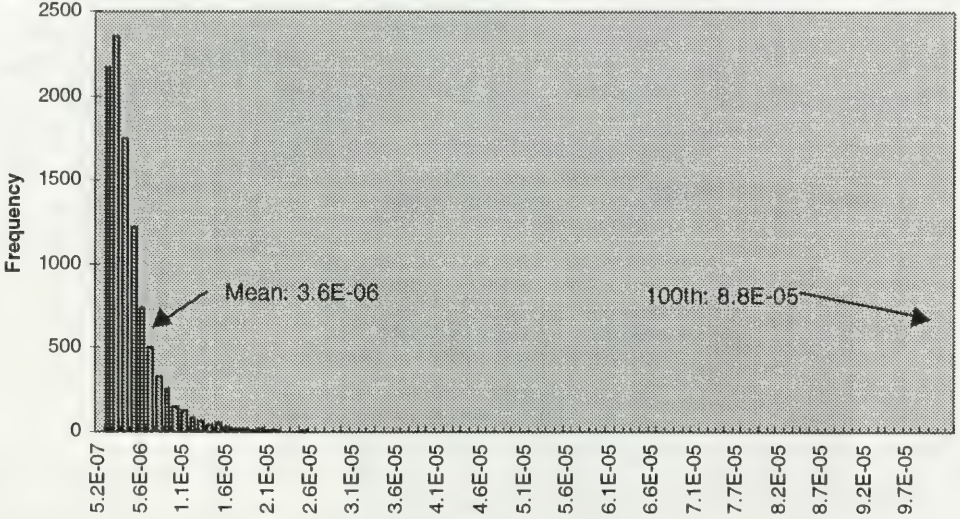


Figure B-8. Distribution of Individual LECR for the Composite Recptor Exposed to the Mixture of Chlorinated Carcinogens at Annual GLCmax during 30 Years

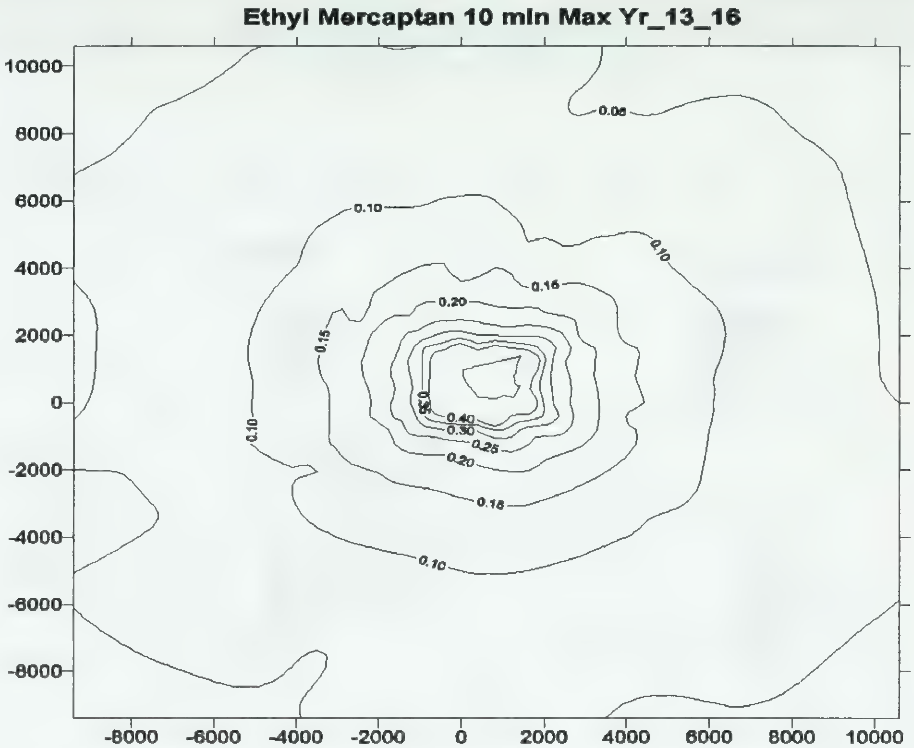


Figure9. Isopleths Describing the Atmospheric Dispersion of the Exposure Ratio (ER) based on the (10m-GLC$_{max}$) of Ethyl Mercaptan for the Emission Period 13-16 years.

Figure B-9. Isopleths Describing the Atmospheric Dispersion of the Exposure Ratio (ER) based on the 10m-GLC$_{max}$ of Ethyl Mercaptan for the Emission Period 13-16 years.

Fiaure 10. Mean % and 97.5th % Contribution of Direct and Secondarv Pathwavs to Fe Intake

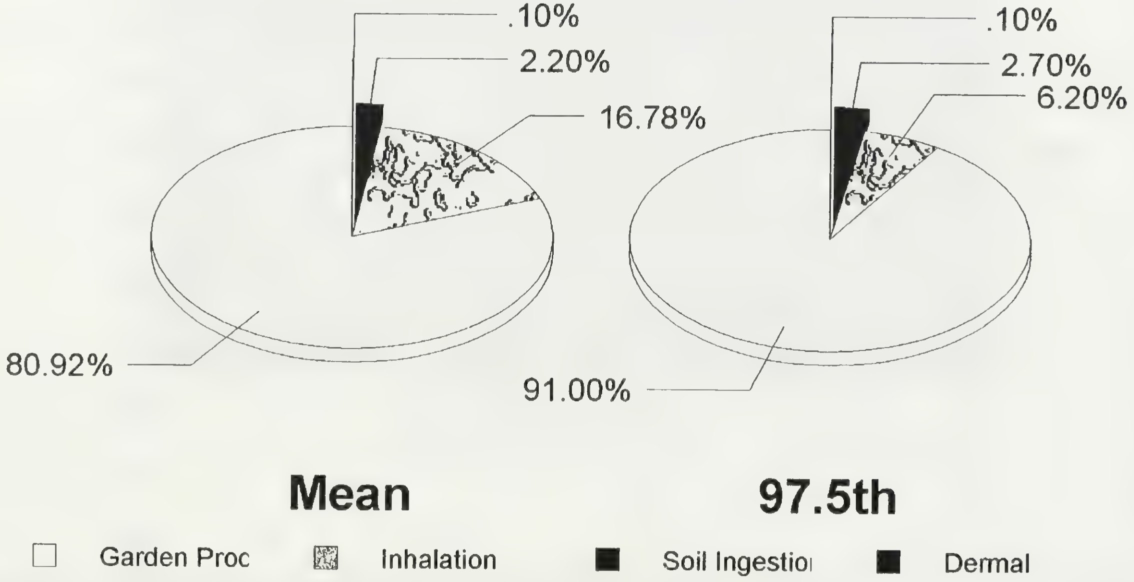


Figure B-10. Mean % and 97.5th % Contribution of Direct and Secondary Pathways to Fe Intake.

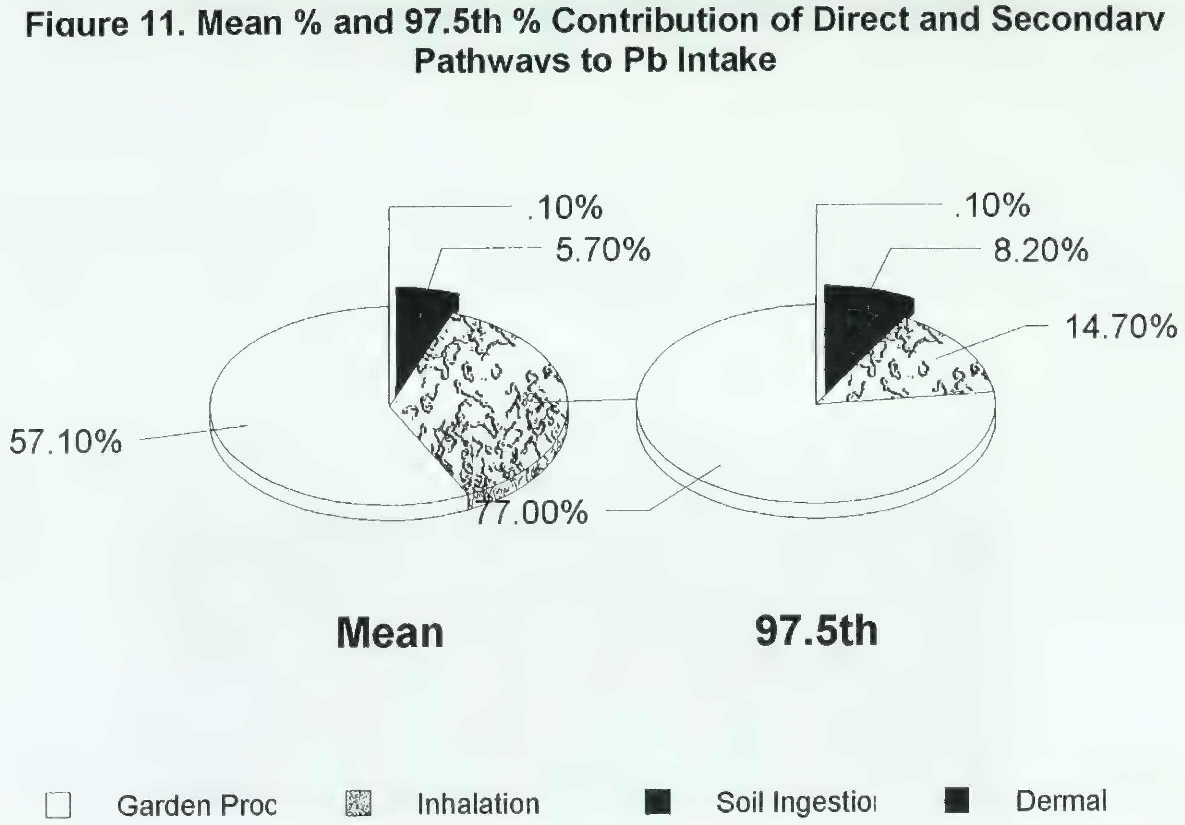


Figure B-11. Mean % and 97.5th % Contribution of Direct and Secondary Pathways to Pb Intake.

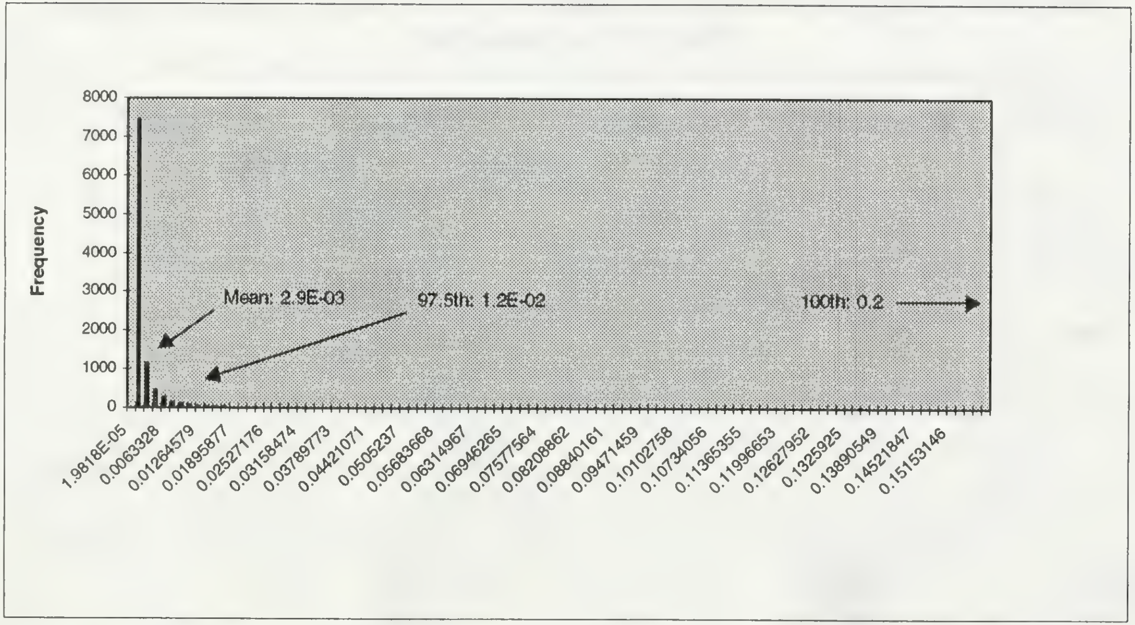
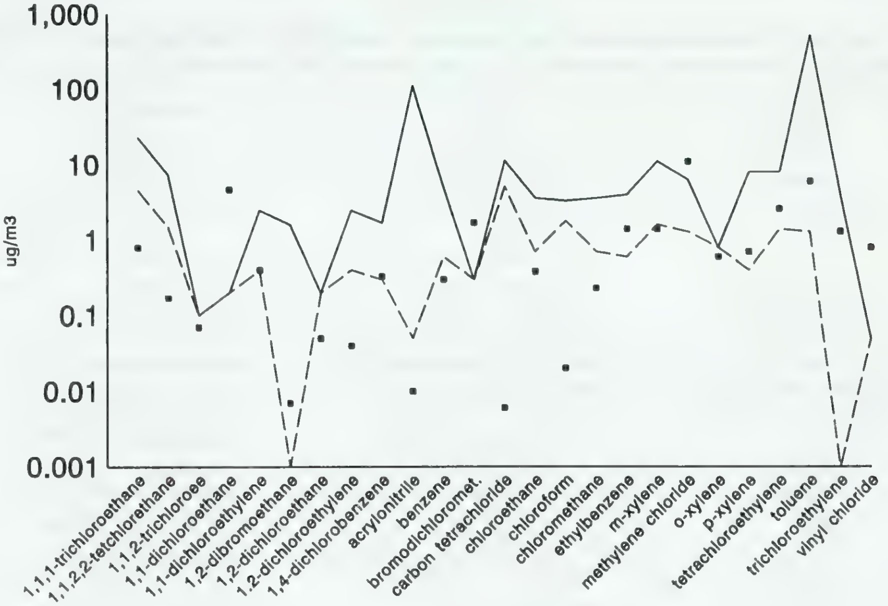


Figure B-12. Distribution of Potential Cases (# people affected) of Adverse Effects among the Population (550,000) Living in the 20 km X 20 km Area Surrounding the Landfill and Exposed to the Chlorinated Volatiles Mixture

Figure 13. Comparison between minimum and maximum background levels with maximum 24h-GLC modelled at year 20 for selected volatiles



minimum background values: broken line (IWA, 1994)
maximum background values: plain line (IWA, 1994)
modelled 24h-GLCmax: points

Figure B-13. Comparison between minimum and maximum background levels with maximum 24h-GLC modelled at year 20 for selected volatiles

Table B-1. Chemicals Detected in Atmospheric Emissions of MSW Landfills in Ontario (IWA, 1994)

1,1-dichloroethane [1]
1,1-dichloroethylene [1]
1,2-dibromoethane [1]
1,2-dichlorobenzene
1,2-dichloroethane [1]
1,2-dichloroethylene [1]
1,3-dichlorobenzene
1,4-dichlorobenzene
2,3-dimethylpentane
3-methylpentane
1,2-dichloropropane
1,1,2,2-tetrachloroethane
1,1,1-trichloroethane [1]
1,1,2-trichloroethane
acetone
acrylonitrile
benzene
bromodichloromethane
butane-2-ol
butane
butylacetate
carbon disulphide

carbon tetrachloride [1]
carbonyl sulphide
chlorobenzene
chlorodifluoromethane
chloroethane
chloroform [1]
chloromethane
cyclohexane
decane

dichlorodifluoromethane
dichlorofluoromethane
dimethyldisulphide
dimethylsulphide
ethyl mercaptan
ethylbenzene
hexane
hydrogen sulphide
m-xylene
methylene chloride [1]
MEK

methyl mercaptan
MIK
nitrogen oxides
nonane
o-xylene
octane
p-xylene
pentane
styrene
tetrachloroethylene [1]
tetrahydrofurane (THF)
toluene
trichloroethylene [1]
trichlorofluoromethane
vinyl chloride [1]

cadmium
chromium
copper
iron
lead
zinc

[1] Chemical indicators identified by MOE (1992)

Table B-2. Model Parameters, Exposure Variables and Scenario-Based Assumptions Used in the Conservative Multi-Media Exposure Model for the Screening of Metals

Variable/Assumption	Definition	Reference/Comment
Soil Mixing Depth (SD)	0.05 m	Assumed for playground setting and gardening
Exposure Period(Tf, Tt) by Inhalation	7300 days (20 years)	Operational lifetime of the landfill
Exposure Period Through Soil-Based Pathways	10950 days (30 years)	Residency time at the impacted location
Rate of outdoor air infiltration (Indoor/Outdoor Ratio)	1	Hawkins, 1991
Exposure from Dermal Contact with Soil	7 days/week x 8 months/year[1]	-
Exposure from Soil Ingestion	7 days/week x 8 months/year[1]	-
Exposure from Ingestion of Garden Produce	7 days/week x 8 months/year[2]	-
Inhalation rate *per* age group	Maximum value reported in IWA (1994)	See Appendix B-II-A
Body weight *per* age group	Average value reported in IWA (1994)	"
Surface area *per* age group	Maximum value reported in IWA (1994) (and Finley *et al.*, (1994) for 0-6m)	"
Soil ingestion rate *per* age group	Maximum value reported in IWA (1994)	"
Leafy crop ingestion rate *per* age group	Maximum value reported in IWA (1994)	"
Vines crop ingestion rate *per* age group	Maximum value reported in IWA (1994) for fruits and vegetables	"
Root crop ingestion rate *per* age group	Maximum value reported in IWA (1994) for potatoes	"
Fraction of garden produce from local origin	leafy: 0.7; vines 0.5; root: 0.2	US EPA, 1990
Fraction of the total mass of ingested fruit and vegetables that consist of unprotected produce	0.47	McKone and Daniels, 1991

[1] Exposure for the 0-6m subgroup was estimated for the 3-6m period under the assumption that no contact with outdoor soil would occur during the first 3 months [2] The 0-6m subgroup was assumed to be breast- and/or formula-fed.

Table B-3. Final List of Chemicals Identified in Ontario Landfill Leachate and Maximum Predicted Concentrations in the Groundwater Aquifer at Receptors Location (adapted from IWA, 1994)

Chemical	Maximum Predicted Concentrations (ug/L)	
	Anticipated Conditions	Reasonable Worst Case Conditions
dichloromethane	8.80E-09	5.00E-02
vinyl chloride	5.50E-05	1.71E-01
1,1-dichloroethylene	2.20E-06	4.20E-02
benzene	8.80E-08	2.60E-02
1,1,2,2-tetrachloroethane	1.80E-12	1.80E-02
bromodichloromethane	2.10E-07	2.80E-04
1,2-dichloroethane	2.20E-05	2.80E-02
pentachlorophenol	1.00E-11	8.10E-07
octanoic acid	2.80E-22	5.00E-03
butyric acid	1.10E-03	9.30E+00
cis-1,2-dichloroethylene	1.00E-02	7.50E-01
dimethyl sulphide	3.90E-07	2.90E-02
o-cresol	1.20E-12	2.20E-04
1,4-dichlorobenzene	1.60E-09	6.80E-03
hexanoic acid	2.60E-07	1.40E+01
chloromethane	2.70E-04	7.00E-02
p-cresol	3.50E-10	5.30E-02
phenyl acetic acid	2.10E-06	6.20E-01
isovaleric acid	7.60E-04	1.10E+01
chloroform	8.70E-07	2.70E-02
chloroethane	1.70E-07	5.90E-03
phenol	1.30E-06	5.50E-01
isobutyric acid	1.43E-03	5.40E+00
1,1-dichloroethane	9.10E-03	4.80E+00
ethyl benzene	1.00E-23	5.50E-04
p- *m*-xylene	1.00E-22	1.60E-03
2,4,6-trichlorophenol	5.50E-08	2.20E-06
toluene	1.50E-11	4.00E-02
trichloroethylene	9.60E-19	4.30E-04
acetic acid	1.90E-04	1.20E+00
propionic acid	7.00E-03	1.60E+01
heptanoic acid	1.30E-17	4.00E-02
bromodichloromethane	7.50E-07	3.10E-02
tetrachloroethylene	2.20E-18	4.00E-04
1,1,1-trichloroethane	5.90E-10	6.50E-02
1,1,2-trichloroethane	5.90E-10	3.00E-02
o-xylene	8.30E-24	3.10E-03
nitrate	9.50E+02	2.00E+03

Table B-4. Model Parameters and Assumptions Used in the Probabilistic Multi-Media Exposure Model Applied to Volatiles

Parameter	Definition	Reference/Comment
Concentrations in Soil		
Annual-GLC$_{max}$ at receptor location (ug/m^3)	LN (SD of 3.5)	OMOE atmospheric modelling staff recommendation
Deposition rate (m/day)	LN (6.33E-03, 9.70E-03) (see Appendix B-II-B)	McKone and Daniels, 1991
Chemical-specific soil Half-life (days)	U (see Appendix B-III-F for VOC's; the CAPCOA default values were used for metals)	Mackay *et al.*, 1992
Soil Mixing Depth (m)	0.05	Assumed value for playground/ gardening
Soil bulk density (kg/m^3)	U (1,000, 1,800) (see Appendix B-II-B)	Copeland *et al.*, 1994
Concentrations in Garden Produce		
Interception fraction for leafy	U (0.15, 0.39) (see Appendix B-II-B)	Copeland *et al.*, 1994
Interception fraction for vines	U (0.03, 0.07) (see Appendix B-II-B)	Copeland *et al.*, 1994
Crop Yield (kg/m^2)	U (2,4) (see Appendix B-II-B)	Copeland *et al.*, 1994
Growth period (days)	U (45, 90) (see Appendix B-II-B)	Copeland *et al.*, 1994
Senescence/weathering (days)^1	LN (10, 3, 2.8, 34) (see Appendix B-II-B)	Copeland *et al.*, 1994
Octanol-water partition coefficients (Kow)	T (see Appendix B-III-F)	Mackay *et al.*, 1992
Organic carbon partition coefficient (Koc)	T (see Appendix B-III-F)	Mackay *et al.*, 1992
Henry's law constants (H)	T (see Appendix B-III-F)	Mackay *et al.*, 1992
Exposure Parameters and Scenario-Based Assumptions		
Total Exposure Period (days)	10950 (30 years)	Upper 95th$_{percentile}$ of residency period reported by Finley *et al.*, 1994
Averaging time (days)	25550 (70 years)	Lifespan for the composite receptor
Body weight *per* age group (kg)	see Appendix B-II-A	IWA, 1994

Table B-4. Model Parameters and Assumptions Used in the Probabilistic Multi-Media Exposure Model Applied to Volatiles

Parameter	Definition	Reference/Comment
Inhalation rate *per* age group (m^3/day)	see Appendix B-II-A	IWA, 1994
Indoor/Outdoor ratio (I/O)	Assumed T (0.2, 0.75, 1.0) (see Appendix B-II-A)	Hawkins, 1991; Hawley, 1985
Fraction of time spent outdoors (T_o) and indoors (T_i)	0.8 - 0.2	Manca *et al.*, 1995
Exposure from Dermal Contact with Soil	7 days/week x 8 months/year (none for first 3 months of 0-6m subgroup)	Scenario-based assumption
Surface area *per* age group (cm^2)	see Appendix B-II-A (active day values from IWA, 1994)	IWA, 1994; Finley *et al.*, 1994
Soil loading factor (mg/cm^2-day)	LN (0.52, 0.90) (see Appendix B-II-A)	Finley *et al.*, 1994
Fraction of skin absorption	0.03	McKone, 1990
Exposure from Soil Ingestion	7 days/week x 8 months/year (none for the first 3 months of the 0-6m subgroup)	Scenario-based assumption
Soil ingestion rate *per* age group (mg/day)	see Appendix B-II-A	IWA (1994)
Fraction of chemical desorbed from ingested particles (ABS)	1	CAPCOA, 1993
Fraction of chemical desorbed from garden produce (ABS)	1	CAPCOA, 1993
Exposure from Ingestion of Garden Produce	7 days/week x 8 months/year (none for the 0-6m subgroup as it is assumed to be breast- or formula-fed)	Scenario-based assumption
Leafy crop ingestion rate *per* age group (kg/day)	see Appendix B-II-A	IWA (1994)
Vines crop ingestion rate *per* age group (kg/day)	Values for fruits and vegetables (see Appendix B-II-A)	IWA (1994)
Fraction of garden produce from local origin	leafy: 0.7; vines 0.5; root: 0.2	US EPA, 1990
Fraction of the total mass of ingested fruit and vegetables that consist of unprotected produce	0.47	McKone and Daniels, 1991

LN (mean, std. dev, min, max): log-normal distribution; U (min, max): uniform distribution; T (min, most likeliest, max): triangular distribution;

[1] for particulate-bound metals

Table B-5. Results of the Screening-Level Assessment - Chemicals Retained for the Detailed Risk Assessment

Carcinogenic Volatiles	Systemic Volatiles	Particulate-Bound Metals	Nuisance Chemicals	Leachate Components
1,1-dichloroethane trichloroethylene 1,2-dichloroethane 1,1,2-trichloroethane chloroethane benzene vinyl chloride bromodichloromethane 1,1-dichloroethylene 1,1,2,2-tetrachloroethane methylene chloride	methyl mercaptan ethyl mercaptan hydrogen sulfide	Fe Zn Pb	ethyl mercaptan methyl mercaptan octane butan-2-ol dimethylsulphide hydrogen sulphide	vinyl chloride

Table B-6a. $LADD_{pot}$ for the Composite Receptor Exposed to Carcinogenic Volatiles at the Annual-GLC_{max}. Deterministic Assessment Using Central-Tendency Values

Chemical	Inhalation $LADD_{POT}$ ug/kg-day	Ingestion $LADD_{POT}$ (ug/kg-day)		Dermal $LADD_{POT}$ ug/kg-day	Total $LADD_{POT}$ *per* chemical ug/kg-day
		Soil	Garden Produce		
Chlorinated Compounds					
1,1,2,2-tetrachloroethane	1.60e-03	3.80e-06	1.20e-06	1.20e-06	1.60e-03
bromodichloromethane	1.60e-02	1.40e-04	1.40e-06	4.60e-05	1.60e-02
1,1-dichloroethane	4.00e-02	3.00e-04	5.80e-07	9.50e-05	4.00e-02
methylene chloride	1.10e-01	2.90e-04	1.50e-06	9.30e-05	1.10e-01
1,1-dichloroethylene	7.20e-03	6.60e-05	8.50e-09	2.40e-05	7.30e-03
vinyl chloride	7.80e-03	6.60e-05	4.10e-09	2.10e-05	7.90e-03
trichloroethylene	1.30e-02	2.90e-05	3.80e-07	9.40e-06	1.30e-02
1,2-dichloroethane	5.40e-04	2.40e-06	2.00e-08	7.80e-07	5.40e-04
1,1,2-trichloroethane	6.60e-04	1.00e-05	1.20e-07	3.40e-06	6.70e-04
chloroethane	3.70e-03	9.80e-06	1.50e-08	3.20e-06	3.70e-03
Others					
benzene	2.60e-03	2.30e-06	8.70e-08	7.30e-07	2.60e-03

Table B-6b. Individual LECR for the Composite Receptor Exposed to Carcinogenic Volatiles (Annual-GLC$_{max}$. Deterministic Assessment - Central-Tendency Values

Chemical	Inhalation LECR	Ingestion LECR		Dermal LECR	Total LECR *per* chemical
		Soil	Garden Produce		
Chlorinated Compounds					
1,1,2,2-tetrachloroethane	2.30e-07	7.70e-10	2.30e-10	2.50e-10	2.30e-07
bromodichloromethane	7.90e-07	8.80e-09	8.70e-11	2.80e-09	8.00e-07
1,1-dichloroethane	1.50e-07	1.70e-09	3.30e-12	5.40e-10	1.50e-07
methylene chloride	1.20e-07	2.20e-09	1.20e-11	7.00e-10	1.30e-07
1,1-dichloroethylene	8.60e-07	3.90e-08	5.10e-12	1.40e-08	9.10e-07
vinyl chloride	1.60e-06	3.50e-08	2.10e-12	1.10e-08	1.60e-06
trichloroethylene	6.20e-08	1.80e-10	2.30e-12	5.70e-11	6.20e-08
1,2-dichloroethane	3.30e-08	2.20e-10	1.80e-12	7.10e-11	3.40e-08
1,1,2-trichloroethane	2.50e-08	5.90e-10	6.60e-12	1.90e-10	2.60e-08
chloroethane	1.60e-08	1.30e-10	2.00e-13	4.10e-11	1.60e-08
	Sub-Total	8.90e-08	3.54e-10		
Total Inhalation: 3.8E-06 (97.0%)[1]		*Total Ingestion: 8.9E-08 (2.3%)*		*Total Dermal: 3E-08 (0.8%)*	*Total per class: 4E-06*
Others					
benzene	*Sub-Total*	*6.50e-11*	*2.40e-12*		
Total Inhalation: 5.1E-08 (99.8%)		*Total Ingestion: 6.8E-11 (0.13%)*		*Total Dermal: 2.1E-11 (0.04%)*	*Total LECR: 5.1E-08*

[1] % contribution to Total LECR

Table B-7. Individual LECR for the Composite Receptor Exposed to Chlorinated Carcinogenic Volatiles at the Annual-GLC_{max}.
Probabilistic Assessment

Route of Exposure	Mean	50th	95th	97.5th
Chlorinated Carcinogens (including vinyl chloride)				
Inhalation	3.60e-06	2.90e-06	7.80e-06	1.00e-05
Ingestion (total)	5.10e-08	1.80e-08	1.80e-07	2.90e-07
Dermal	2.40e-08	4.80e-09	8.60e-08	1.60e-07
Total LECR	3.60e-06	2.90e-06	7.90e-06	1.00e-05
Vinyl Chloride				
Inhalation	1.40e-06	9.50e-07	4.10e-06	5.60e-06
Soil Ingestion	1.70e-08	6.30e-09	6.30e-08	1.10e-07
Ingestion of Garden Produce	1.80e-13	7.00e-14	6.50e-13	1.10e-12
Dermal	9.10e-09	6.50e-09	3.40e-08	5.70e-08
Total LECR	1.50e-06	9.70e-07	4.20e-06	5.70e-06

Table B-8. Individual Exposure Ratios (ER) Determined for Non-Carcinogenic Systemic Volatiles during the Period of Maximum Emissions at the Point of Maximum Impact - Probabilistic Assessment

Emission Period/Chemicals	Mean Annual-GLC_{max} ug/m^3	Mean ER	50th ER	95th ER	97.5th ER
17-20 years					
Hydrogen sulfide	0.011	0.01	0.005	0.04	0.07
Ethyl mercaptan	0.18	0.18	0.07	0.7	1.1
Methyl mercaptan	0.27	0.26	0.08	1	1.7

Table B-9. Individual Exposure Ratios Determined for Volatiles Associated with Odor Effects[1] at the Point of Maximum Impact Based on 10m-GLC Deterministic Assessment

Emission Period	ethyl mercaptan	methyl mercaptan	octane	butan-2-ol	dimethylsulphide	hydrogen sulphide
Year 1-4						
ER	(6.0) 6.7	(4.1) 1.7	(7.7) 0.12	(25.8) 0.04	(2.4) 0.05	(0.24) 0.04
Year 5-8						
ER	(3) 3.3	(4.4) 1.8	(10.5) 0.2	(35.2) 0.06	(1.8) 0.03	(0.2) 0.03
Year 9-12						
ER	(12.5) 14	(19) 8	(33) 0.5	(112) 0.2	(7) 0.2	(0.8) 0.14
Year 13-16						
ER	(24.2) 27	(35) 15	(62) 1	(206) 0.4	(14) 0.3	(1.4) 0.3
Year 17-20						
ER	(11) 12	(17) 7	(28) 0.5	(94) 0.2	(6) 0.13	(0.7) 0.12
Year 21-25						
ER	(16) 18	(24) 10	(42) 0.7	(141) 0.2	(10) 0.2	(1) 0.17
Year 26-30						
ER	(4.5) 5	(6.7) 3	(11.7) 0.19	(39) 0.7	(2.7) 0.05	(0.28) 0.05

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[1] Values in brackets are the corresponding 10m-GLC$_{max}$ in ug/m^3 estimated by multiplying 1h-GLC$_{max}$ by a factor of 2)

Table B-10. Route-Specific ADD_{pot}, Total ADD_{pot}, and ER Associated with Particulate-Bound Elements of Potential Health Concern - Deterministic Default Values

Element	ADD_{pot} Inhalation	ADD_{pot} Ingestion (ug/kg-day)		ADD_{pot} Dermal	Total ADD_{pot}	Total ER
	ug/kg-day	Soil	Garden Produce	ug/kg-day	ug/kg-day	
Iron (Fe)						
Mean	8.4E-02	1.1E-02	4.0E-01	4.6E-04	4.9E-01	1.2
50th	8.3E-02	5.3E-03	2.1E-01	1.3E-04	3.0E-01	0.7
95th	1.2E-01	3.6E-02	1.4E+00	1.8E-03	1.5E+00	3.7
97.5th	1.3E-01	5.5E-02	1.9E+00	3.0E-03	2.0E+00	5.2
Zinc (Zn)						
Mean	3.7E-02	4.7E-03	1.8E-01	2.0E-04	2.2E-01	0.2
50th	3.7E-02	2.4E-03	9.3E-02	5.7E-05	1.3E-01	0.1
95th	5.1E-02	1.6E-02	6.0E-01	7.9E-04	6.6E-01	0.7
97.5th	5.4E-02	2.4E-02	8.4E-01	1.3E-01	9.0E-01	0.9
Lead (Pb)						
Mean	2.0E-02	3.1E-03	3.1E-02	4.2E-05	5.4E-02	0.03
50th	1.9E-02	1.4E-03	1.5E-02	1.2E-05	3.7E-02	0.02
95th	2.9E-02	1.1E-02	1.1E-01	1.6E-04	1.4E-01	0.08
97.5th	3.1E-02	1.7E-02	1.6E-01	2.7E-04	1.9E-01	0.10

Table B-11. Individual LECR for the Composite Receptor Exposed to Chlorinated Carcinogenic Volatiles Present in the Groundwater Aquifer Under Anticipated Conditions
Deterministic Assessment

Chemical	ADD$_{pot}$ (ug/kg-day)					LADD$_{pot}$	LECR
	0-6m	7m-4y	5-12	13-20	21+	ug/kg-day	
vinyl chloride	7.00e-06	3.50e-06	2.10e-06	1.10e-06	1.40e-06	7.70e-07	4.00e-10

Table B-12. Probabilistic Risk Estimates for the Entire Population Living in Proximity of the Hypothetical Landfill[1]

Chemical	Population Risk (or Theoretical Number of People Affected[2])			
	Mean	50th	95th	97.5th
1,1,2,2-tetrachloroethane	6.00e-06	2.00e-06	3.00e-05	4.00e-05
bromodichloromethane	2.00e-04	9.00e-05	8.00e-04	1.00e-03
1,1-dichloroethane	2.00e-03	1.00e-03	9.00e-03	1.00e-02
methylene chloride	2.00e-04	2.00e-04	9.00e-04	1.00e-03
1,1-dichloroethylene	< 0.01	< 0.01	< 0.01	<0.01
vinyl chloride	2.00e-04	6.00e-05	8.00e-04	1.00e-03
trichloroethylene	1.00e-05	3.00e-06	5.00e-05	8.00e-05
1,2-dichloroethane	3.00e-07	8.00e-08	1.00e-06	2.00e-06
1,1,2-trichloroethane	3.00e-07	8.00e-08	1.00e-06	2.00e-06
chloroethane	1.00e-06	5.00e-07	4.00e-06	7.00e-06
Total for the Chlorinated Mixture	3.00e-03	2.00e-03	1.00e-02	2E-02 (100th: 6E-01)

[1] Based on an assumed population of approximately 550,000 living within a 20 km x 20 km area around the landfill

[2] For practical purposes, the number in this table should be interpreted as meaning "0 cases projected for the entire area"

Table B-13. Summary of the Major Assumptions Used in the Probabilistic Assessment
▸ Upper 95th confidence limits were used for cancer potency factors ▸ Uninterrupted residency period of 30 years at the same location ▸ Residence located at the point of maximum impact, *i.e.*, the property line (not applicable to the computation of population risk) ▸ Consumption of garden produce and contact with contaminated soil occurs during 7 days/week x 8 months/year x 30 years ▸ 100% desorption assumed for chemicals present in ingested garden produce and soil particles ▸ no provision for estimating the deposition pattern of particulate material in the respiratory tract when assessing particulate-bound metals ▸ Indefinite half-life of metals in soil ▸ Absence of atmospheric decay in the case of organics ▸ Concentrations of particulate-based metals corresponding to breathing-zone levels measured on-site

Table B- 14. Concentrations of Chemical Classes Measured by Deloraine *et al.* (1995) *vs* the Maximum 24h Ground-Level Concentrations Predicted at Receptor Location at Year 20 (24h-GLC$_{max}$)

Chemical Class	Deloraine *et al.* (1995)	Present Analysis
	100 m Downsite	24h-GLC$_{max}$ at receptor location and year 20
	ug/m^3	
Saturated aliphatic hydrocarbons	33.8	28.2
Unsaturated aliphatic hydrocarbons	19.4	-
Saturated alicyclic hydrocarbons	4.7	-
Unsaturated alicyclic hydrocarbons	1.4	-
Saturated aromatic hydrocarbons	302	4.4
Unsaturated aromatic hydrocarbons	19.4	5.6
Halogenated hydrocarbons	17.8	48
Polycyclic aromatic hydrocarbons	14.3	-
Nitrogen compounds	5.1	4.8
Sulphur compounds	0.6	8.0
Oxygen compounds	13	20
Miscellaneous	1.8	-
Total Maximum	433.4	119

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TECHNICAL REPORT – PART “B”

ASSESSMENT OF THE POTENTIAL HEALTH RISKS ASSOCIATED WITH CHEMICAL EMISSIONS FROM MODERN MUNICIPAL SOLID WASTE LANDFILLS IN ONTARIO

APPENDICES B-I TO B-III

APPENDIX B-I Modelling Emission Rates and Air Concentrations Due to a Landfill and Population Risk around a Landfill.

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B-I-1.0 Site Description

The site chosen for the study has a capacity of 6.6 million tonnes with a life of 20 years. The annual waste deposition rate is presented in Table B-I-1. The site is expected to have a landfill gas (LFG) collection and enclosed flare control system by the end of year 4. The collection system during the active landfilling period will consist of a network of horizontal trenches, supplemented by vertical wells in areas that have reached their final contour.

Figure B-I-1 presents the site plan of the landfill. Waste is deposited in different areas of the landfill over the years. Waste from years 1-4 is deposited in the area identified as Area 1. Waste from years 5-8 is deposited in the area identified as Area 2. Waste from years 9-20 is deposited in the area identified as Area 3. The total landfil area is about 800 m by 700 m.

Annual average emission rates from the landfill were estimated at various stages in the lifetime of the landfill and this data was used as input to a dispersion model used to calculate maximum hour average, maximum 24 hour average and annual average concentrations around the landfill. Emission estimates and model runs were made for 8 time segments extending to 40 years after the landfill opens. Each of the 3 landfill areas was subdivided into 7 to 13 sections to allow more accurate modelling of the landfill surface gas emissions. Emissions from the flare stack also varied over the landfill lifetime. These emissions were modelled using the flare buoyancy appropriate for each time segment. Air concentrations were calculated at a set of receptor locations extending from the property boundary out to 10 km from the landfill.

B-I-1.1 Landfill Gas Emissions

Landfill gas generation was estimated by the USEPA's Landfill Air Emissions Estimation Model (LAEEM), which is a first order decay model as follows:

$$Q = k\ Lo \sum_{i=1}^{n} M_i \exp(-k\ t)$$

where:

$Q =$ landfill gas methane generation rate (scfm)
$Lo =$ methane generation potential (ft^3/tonne)
$k =$ methane rate constant ($year^{-1}$)
$M_i =$ Mass of refuse in the i^{th} year (tonnes)
$t =$ time (years)

The Lo and k constants used are as follows:

$Lo = 4410\ ft^3/tonne$
$k = 0.04\ yr^{-1}$

The k value was considered to be more appropriate for the wetter Ontario landfills. The Lo value is considered to be the median value typical of Ontario sites. No lag period was used in the LAEEM.

B-I-1.1.1 Landfill Gas Collection and Emissions

The LFG generated was assumed to be released by one of two pathways:

- landfill surface; or
- enclosed flare system.

The LFG generation rate and collection efficiency over 40 years is presented in Table B-I-2. The LFG collection system is not expected to be in place until the end of year 4, therefore, no LFG will be collected until the start of year 5. LFG collection efficiency is expected to be approximately 50% before the entire landfill surface is covered by final cover. After the site has reached its capacity at the end of year 20, the application of final cover is expected to be completed with one year. LFG collection efficiency is expected to reach 70% following the completion of final cover. Final cover is expected to be a silty-sand material. Table B-I-2 presents the total LFG (methane and carbon dioxide) emissions from the landfill surface and the LFG collected by collection system and pumped to the flare. Only years 4, 8, 12, 16, 20, 25, 30 and 40 were examined.

B-I-1.1.2 Non-methane Organic Compounds (NMOCs)

A number of NMOCs are found in LFG. These concentrations of NMOCs within the landfill can be used in the LAEEM to estimate the individual NMOC emission rates. A list of 21 NMOCs was chosen for further evaluations, based on screening risk analysis.

There are several sources of NMOC concentration available in the literature. USEPA and California Air Resources Board (CARB) have compiled extensive data on NMOC concentrations within landfills. In Ontario, data is available for Britannia, Keele Valley, Essex No. 3, Beare Road, Brock West and Eastview landfills. The Britannia data was the most comprehensive, therefore, it was chosen for use in this study. Where data was not available from Britannia or other Ontario landfills, USEPA data was used to fill in the missing data. The concentration of NMOC used in emission estimates are present in the third column from the right in Table B-I-3.

The NMOCs concentrations are combined with the LAEEM results to derive NMOC emission rates. NMOCs were assumed to be present at the concentrations in the third column from the right in Table B-I-3, in both the LFG surface emissions and the LFG sent to the enclosed flare. The enclosed flare was assumed to have a control efficiency of 98% for each individual NMOC. The NMOC landfill surface emission rates from the three areas of landfill waste deposits are presented in Tables 4a to 4f. The surface emissions from Areas 1, 2 and 3 were estimated separately to facilitate dispersion modelling.

B-I-1.1.3 Variability in NMOCs

There are variations in the NMOC concentrations within the same landfill, and from one landfill to another landfill. The concentration of NMOC concentration in the landfill gas is directly proportional to the modelled concentrations downwind of the landfill from both the flare and the landfill surface.

Variability was examined by reviewing the available median concentration information from six Ontario landfills and one set of USEPA data (Table B-I-3). Landfill gas concentrations of NMOCs were examined and reviewed to select a subset of contaminants with more than 2 data points. This subset included: vinyl chloride, 1,1-dichloroethane, 1,1-dichloroethylene, methylene chloride, trichloroethylene, 1,2-dichloroethane, chloroform, chloromethane, hydrogen sulphide and benzene. The geometric mean and geometric standard deviations (GSD) were calculated for each chemical in the subset. A lognormal distribution was assumed since environmental monitoring data often follows this type of distribution. The average GSD for the subset was 4.4, the median GSD was 3.7. If the two highest and two lowest GSDs were eliminated, then the average GSD would drop to 3.7. A value of 3.7 was considered most appropriate for the general NMOC data for this study. This GSD of 3.7 was then applied to the values in the third column from the right to derive the 5th and 95th percentiles (Table B-I-3).

B-I-1.1.4 Enclosed Flare Emissions

The enclosed flare was designed to burn the LFG at a temperature of 1800°F with a residence time of greater than or equal to 1 second. The flare was designed for the maximum gas flow year, which is year 21 just after the completion of final cover. A heat and mass balance was conducted for each of the years examined. An enclosed flare 1.8 m (6 ft) in diameter and 10.4 m (34 ft) in height was considered to be the minimum needed to achieve the design temperature and residence time. The combustion gas flow rate, stack velocity and temperature, for each of the years examined, are presented in Table B-I-5. The NMOC emission rates are presented in Table B-I-6.

B-I-1.2 Other Modelling Data Files

Since annual average air concentrations are one of the outputs required from the dispersion modelling, model runs were performed using a year long hourly meteorological data set. Other than meteorology and emission files, the other 2 data sets needed to perform dispersion calculations are a source characteristics file and a set of receptor locations at which concentrations are calculated.

B-I-1.2.1 Meteorological Data

To run a dispersion model on an sequential hour by hour basis, meteorological information is needed on the 10 m wind speed and direction, the air temperature, the fractional cloud coverage or opacity and the depth of the convective mixing layer where appropriate. In addition information on ground snow coverage and on the roughness of the ground surface is required by the modelling package used in this study. The information on wind speed and direction as well as air temperature can be obtained for most airports and at other monitoring networks including those maintained by MOEE. Hourly cloud coverage over the entire day is however only available at the larger airport which are operated throughout the day. To obtain convective mixed layer heights temperature profile data obtained from upper air instrumented balloon releases are needed. This information is only available at specific airport or special sites in Canada and the US.

Wind and temperature data from Buttonville Airport located north of Metropolitan Toronto were selected as the most applicable to this assessment. Since wind speeds at the MOEE tower in Oshawa were generally lower than would be expected at the mainly rural Durham Landfill site, these data were not as representative as the Buttonville data. However, since Buttonville does not have nighttime cloud coverage information was obtained from Toronto International Airport. Snow coverage was based on daily maps giving the 1 and 6 inch snowlines across North America. These data were combined with convective mixed layer heights obtained using surface temperatures and the morning upper air sondes released at Buffalo. The methodology described in Holtzworth (1967) was used to calculate the daytime mixing heights. The meteorological data set assembled was for one year (1989).

Figure B-I-2 show the wind rose and the wind speed frequency distribution for the data set. The most frequent wind directions are from the north north west and from the west south west.

B-I-1.2.2 Receptor Locations

Calculations were performed for nested grids extending from the landfill property line out to 10 km from the landfill. No calculations were performed on the landfill site itself. Figure B-I-3 shows the locations of the model grid points. Along the property line receptors were located at about 60 m intervals. A grid resolution of 250 m was used out to 2 km with a 500 m resolution from 2 to 4 km and a 2000 m resolution from 4 to 10 km. The total numder of model receptors was 614.

B-I-1.2.3 Source Characteristics

The three landfill deposit areas shown in Figure B-I-1 were subdivided into sections about 100 to 150 m across. This allows the modelling to more accurately reflect the uniform emissions assumed to occur across each of the three landfill areas. An initial total vertical mixing height of 2 m was used for each of the 29 subdivided area sources in the modelling assessment. For each of the 8 time periods over the landfills lifetime, the emission rates per m^2 for the three landfill deposit areas given in Tables B-I-4 a-f are assumed to be uniform over all of the model subdivisions within each landfill area. The emission rates are then proportional to the area of each subdivision used.

The flare stack is located south of the landfill as shown on Figure B-I-1. Table B-I-5 lists the exit gas velocities of the flare for the various stages in the landfills lifetime. The enclosed flare stack is 10.2 m high with a stack diameter of 1.8 m. The gas temperature as it exits the stack is 980 °C.

B-I-1.3 Modelling Method

The dispersion model used in this study is the CAP model developed by MOEE (Appendix 8, Clean Air Program(CAP)). The CAP modelling package consists of different modules for use in specific meteorological conditions. The general categories of meteorological conditions covered by the CAP model are stable (i.e., usually clear skies at night with light winds), neutral (windy and cloudy), convective (daytime with thermal up and down drafts created by the sun heating the earth surface) and lakeshore fumigation (during warm season days near large lakes when a lakebreeze transports stable air from over the lake to the convective airmass over the land). In this modelling package there is a continuous variation of stability as opposed to discrete stability classes. Mechanical mixing is generated by the wind flowing over the surface. The mixing rates depend on the surface roughness. A value of 20 cm was used for the surface roughness in this study.

To evaluate both chronic and acute impacts due to emissions from the landfill, concentrations were needed over the model domain for the annual average, the maximum 24 hour average and the maximum hour average. All three concentration averages were obtained at each model receptor location. The annual average concentrations depend significantly on the wind rose while the maximum hourly average would be more uniformly distributed with respect to wind direction.

Since the emission rates of the NMOCs are a fixed fraction of the gas flow rates from the landfill (i.e., see section B-I-1.1.2) model calculations can be done on a 'unit emission' basis for each of the 8 time periods over the landfills lifetime. The 'unit emission' calculated concentrations can then be scaled by the emission rate per m^2 for each compound of concern. For the emissions through the landfill surface, the ratio's of emissions from the three landfill deposition areas can be obtained from Tables B-I-4b, 4d and 4f. Using area 1 as the baseline, emission rates per m^2 for areas 2 and 3 were then derived as a ratio with respect to area 1. The total emission rate is proportional to the area of each subdivision used in the modelling. Similarly the flare stack emission rates were scaled to the area 1 emission rates to allow a 'unit emission' calculation to be performed simultaneously for all sources.

B-I-1.4 Results

Figure B-I-4 shows contours of annual average concentration patterns for 1,1-dichloroethane (which was used for air modelling) at each of the 8 periods in the landfills lifetime. Over the 10 km region, modelled annual average concentrations distinctly show the predominant north north west wind direction in the meteorological data set. The maximum concentrations are along the landfill property line with the overall maximum located on the southern side for first two periods when waste is deposited in areas 1 and 2 only. As the emissions from area 3 begin to dominate in time period 4, the maximum shifts to the south west property line. The lowest annual average concentrations located in the north west and south west corners of the model domain are about three orders of magnitude smaller than the concentration maximum at the landfill boundary. Calculated air concentrations are a maximum at year 20. After this time the gas collection system is assumed to process 70% of the gas generated by the landfill (i.e., Table B-I-2) which results in a significant drop in overall emission rates.

Figures B-I-5 and B-I-6 depict the maximum 24 hour average and the maximum hour average concentrations respectively. These maxima were selected from the year long meteorological data set at each individual receptor location. The plot for 24 hour average maxima still shows that the wind direction frequency plays a part in the distribution of concentrations although not quite as much as for the annual average. However, the maximum hourly average concentration occurs when the worst dispersion conditions prevail (light wind speeds and stable conditions are worst case for area source emissions). These conditions can occur with any wind direction but will occur more frequently for the predominant wind directions. The highest 24 hour average concentration was about 14 times as large as the highest calculated annual average concentration. For the hourly average, the highest value was about 38 times the highest calculated annual average concentration.

To derive concentrations for any other compound, the results in Figures B-I-5, 6 and 7 are simply multiplied by the emission ratio of the compound of concern relative to 1,1-dichloroethane.

B-I-2.0 Population Risks Around a Landfill

Calculating the probability of an individual developing cancer due to emissions from a landfill requires the following steps.

- The emission rates for each compound are required to estimate air concentrations around the facility. Stack characteristics are also required to perform dispersion modelling. The information used in this study is described in section B-2.2.

- The risk per ug/m^3 for each compound for all appropriate exposure pathways must first be obtained. The procedures to estimate risks and the uncertainty in those risks are described in Section B-2.3.1.4 of this report.

- The variation in emission rates over the landfill area for each compound for the 30 year exposure period used in the assessment is required to estimate air concentrations around the site. A description of the USEPA's Landfill Air Emission Estimates model and the data sources used to estimate emission rates and the uncertainty in those emissions is given in this appendix (section B-I-1.1).

- A dispersion model is then run using a year long hourly meteorological data set to calculate annual average concentrations over a 20 km by 20 km area around the landfill at different stages in the lifetime of the landfill. These yearly concentrations can then be averaged to give a 30 year average exposure concentrations for the area around the landfill. Earlier sections of this appendix include a description of the methodology used to calculate concentrations due to landfill emissions.

- Information on the best estimate and the uncertainty in each of the above factors is then used to estimate risk distributions for each compound at the point of maximum concentrations or at any other location around the landfill. This is done for all compounds of concern.

To estimate the maximum likely population risks around a landfill, the landfill is assumed to be surrounded by a large urban population. For example, the Britannia Landfillin Mississauga has a population of approximately 550,000 living within 10 km of the site. Since the population distribution around a site can vary and the patterns of concentrations for a landfill and an incinerator are different, the fairest estimation of the relative population risks due to the two operations requires that the population distribution be treated statistically. The steps in calculating a population risk along with an uncertainty estimate depends on where the population is located relative to the facility are:

- Use census tract data for the area within 10 km of the Britainnia Landfill to derive a distribution of population densities averaged over 2 km by 2 km grid cells. Given the very large population around this landfill, this provides a maximum likely population risk. Figure B-6 shows the bimodal distribution of population densities derived from the census tract data. The low density mode corresponds to mainly industrial or mainly rural areas while the high density mode represents various types of residential development.

- The above population density data is inputted to create a customized distribution curve using the Crystal

Ball Software.

- Air concentrations for one compound, 1,1-dichloroethane, are derived as 2 km square averages over a 20 km by 20 km area around the landfill. These concentrations are then averaged over the 30 year exposure period. Since the pattern of concentrations are the same for all compounds in this study, calculations for one compound can be used to derive the necessary information to calculate population risks for any other compound.

- Using, a Monte Carlo sampling procedure, a population density is selected from the distribution curve for each of the 100 grid cells for which concentration data is available. The selected population in each cell is multiplied by that cells concentration and the results are then summed over the 20 km by 20 km domain. The final result is a value for population weighted exposure to concentrations of a given compound (i.e., units are Population*ug/m^3).

- The above calculation can then be repeated for many random selections (≈ 100 cases are used) of population densities around the landfill resulting in a series of possible population exposures to concentration. Figure B-7 shows the results of this analysis technique for population exposure to 1,1-dichloroethane. The data are nearly normally distributed.

- Multiplying the individual risk per ug/m^3 for each appropriate exposure pathway for the compound by the population concentration exposure gives the number of people affected over the 30 year period. Since both of these factors have a distribution associated with them, the final answer would be a distribution for the number of people affected. This distribution would represent the combination of all the uncertainties and variabilities in the overall assessment. This includes the uncertainties in the risk estimates for the compound, the variation in responses of individuals to the effect, variations in emission estimates from landfill facilities, and the variation due to the location of population centres relative to the landfill location.

- Having derived population risk numbers for one compound, the risks for other compounds are readily calculated since the risks all vary linearly with concentration and the pattern of concentrations is the same for all compounds.

B-I-3.0 References

USEPA, 1990. Landfill Air Emissions Estimation Model, User's Manual, EPA-600/8-90-85a, December 1990.

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Holtzworth,G.C., 1967, Mixing depths, wind speeds and air pollution potential for selected locations in the United States. J. Applied Met., 6, 1039-1044.

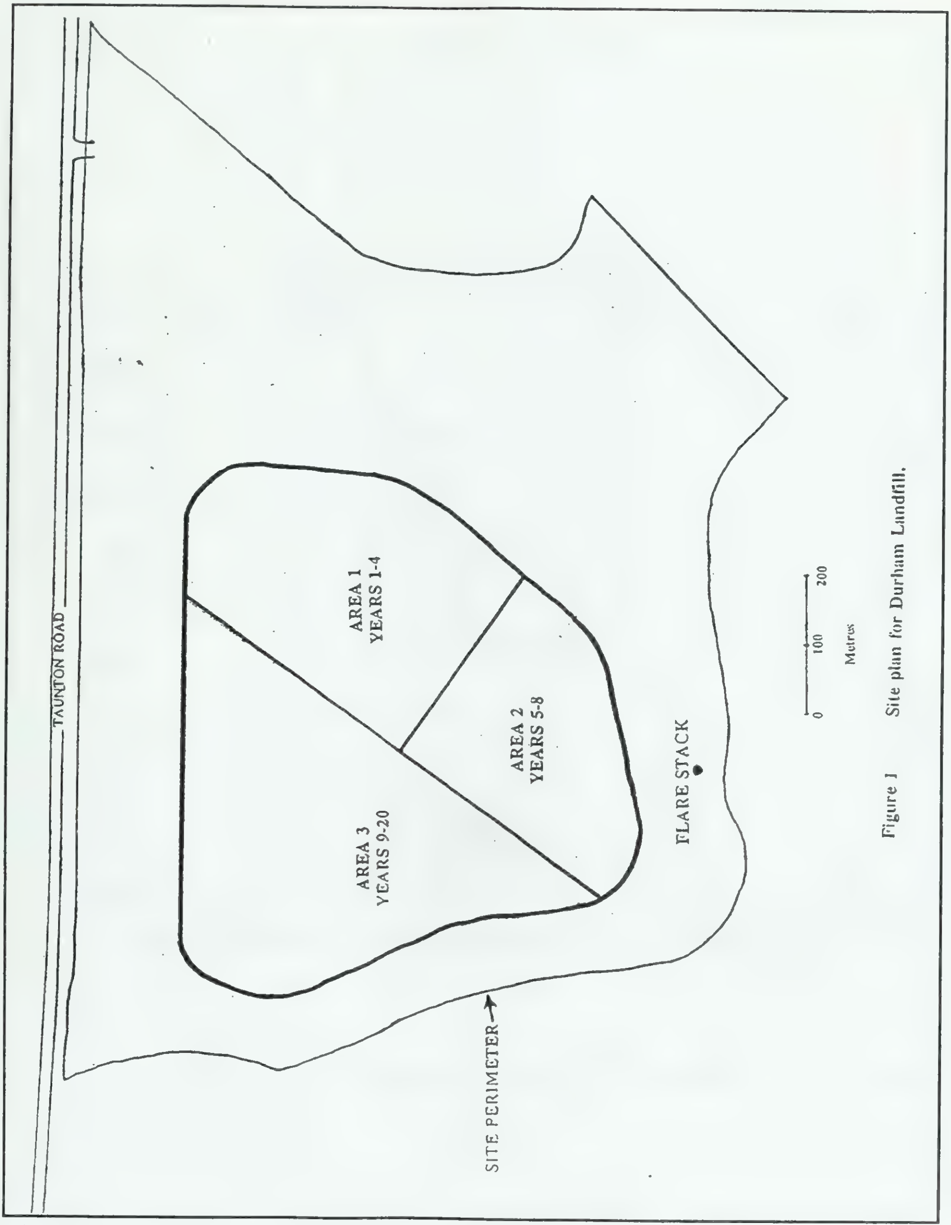


Figure B-II-1. Site Plan for proposed Durham Landfill.

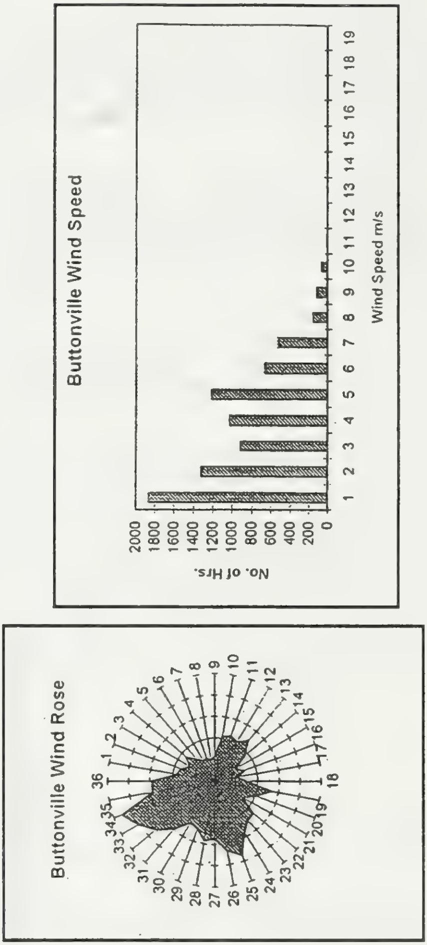


Figure A-II-1 Wind rose and wind speed frequency distribution for Buttonville Airport.

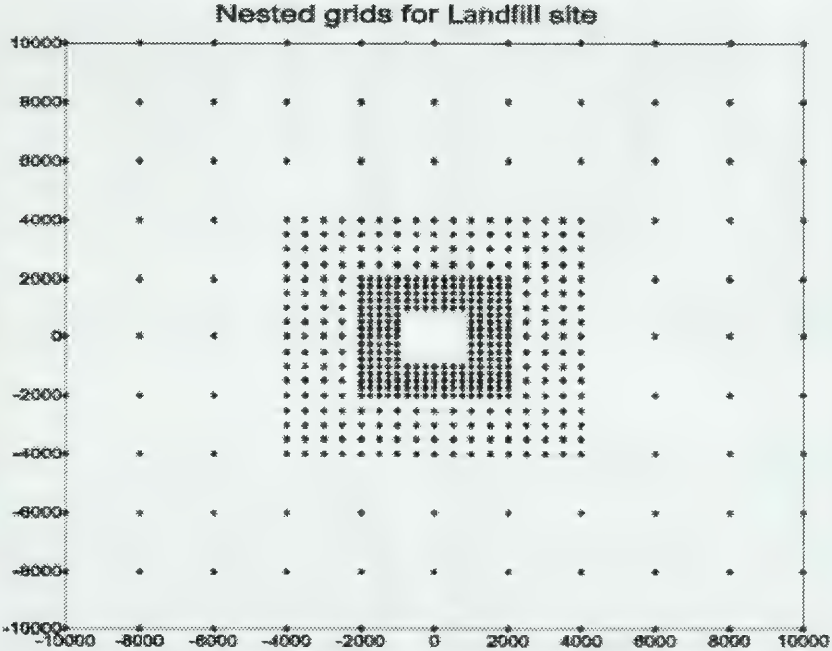
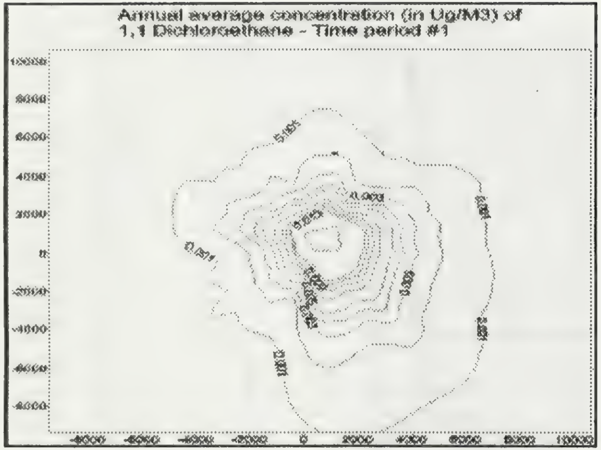
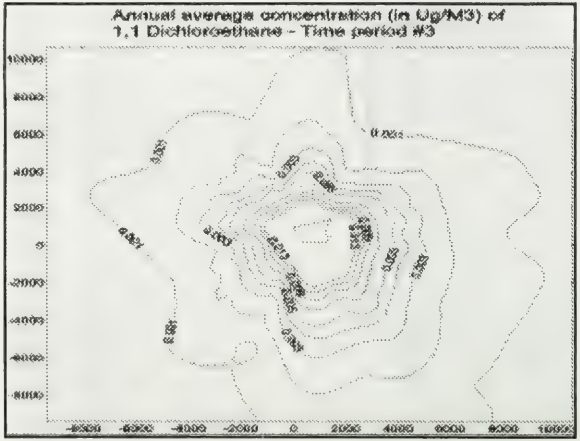


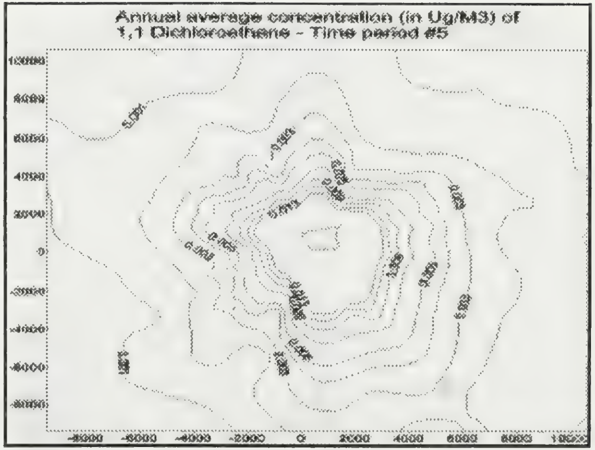
Figure 3: Model receptor locations for a 20 km by 20km grid



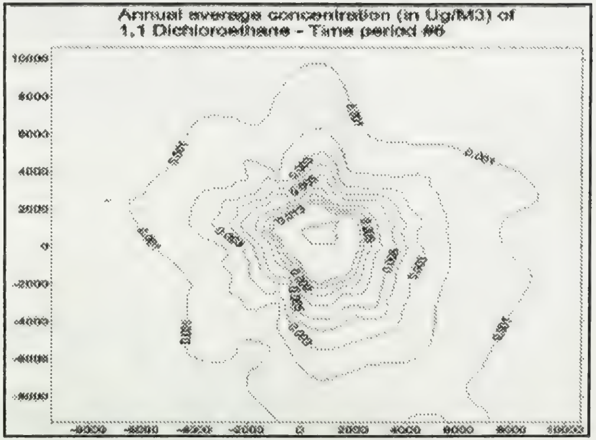
Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #1



Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #3



Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #5



Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #6

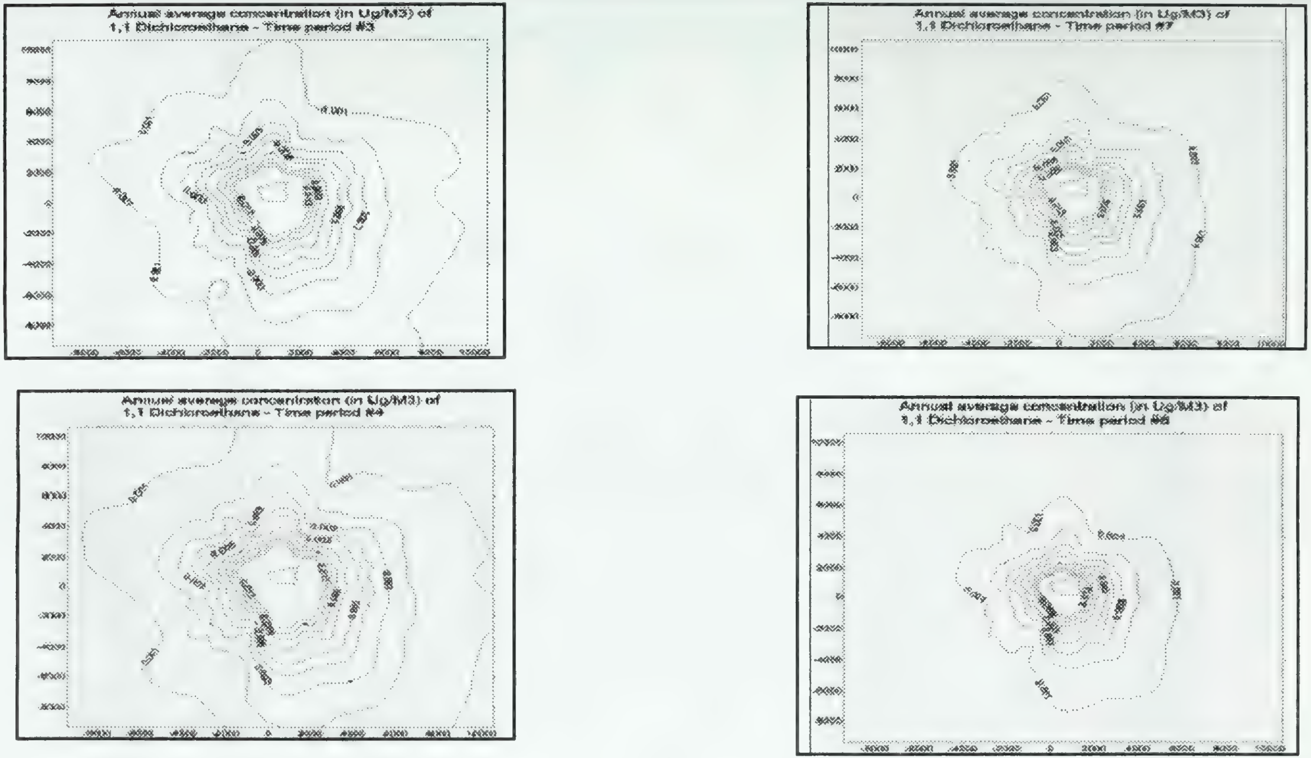

Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #3
Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #7
Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #4
Annual average concentration (in Ug/M3) of
1,1 Dichloroethane - Time period #8
0

Figure B-I-4. Modelled Annual Average Concentrations (ug/m^3) for 1,1-dichloroethane (all eight periods).

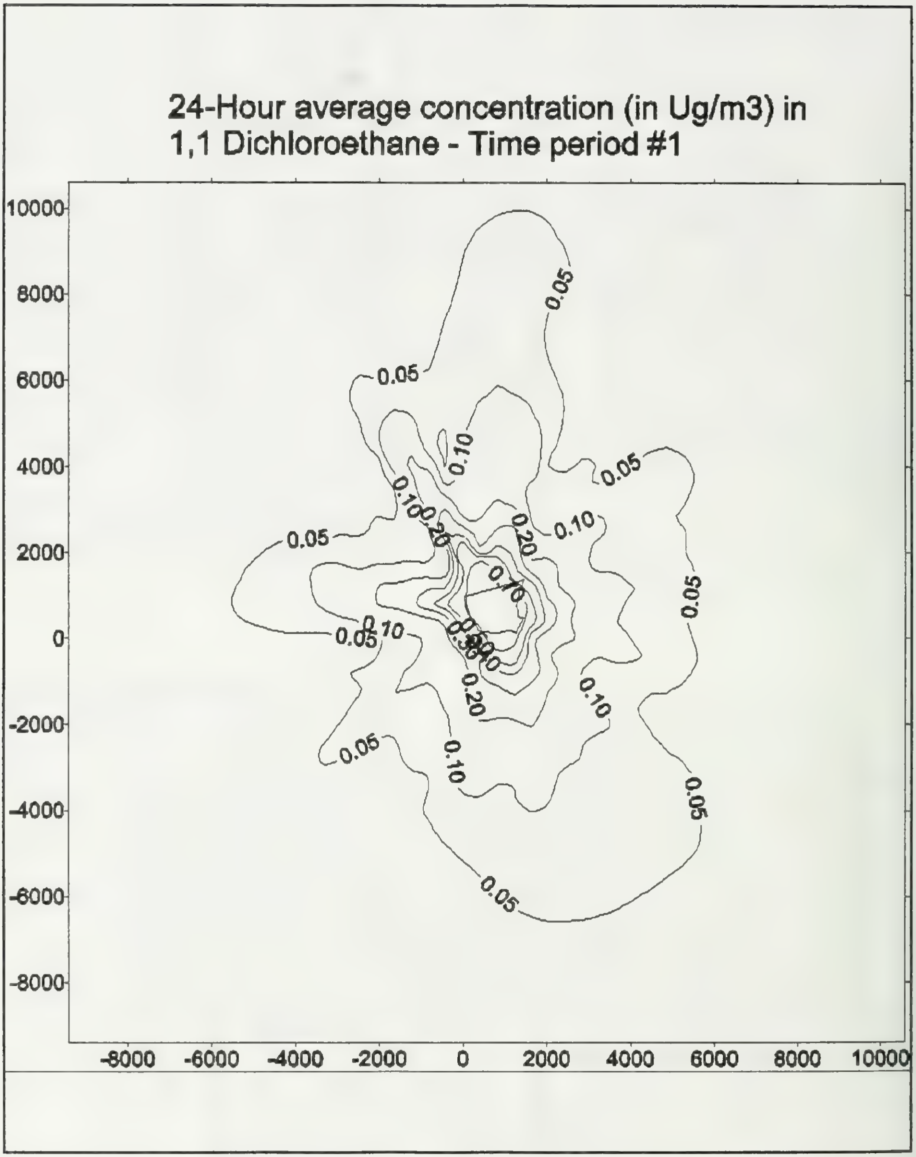


Figure B-II-5. Modelled maximum 24 hour average concentrations for 1,1-dichloroethane (Period 1).

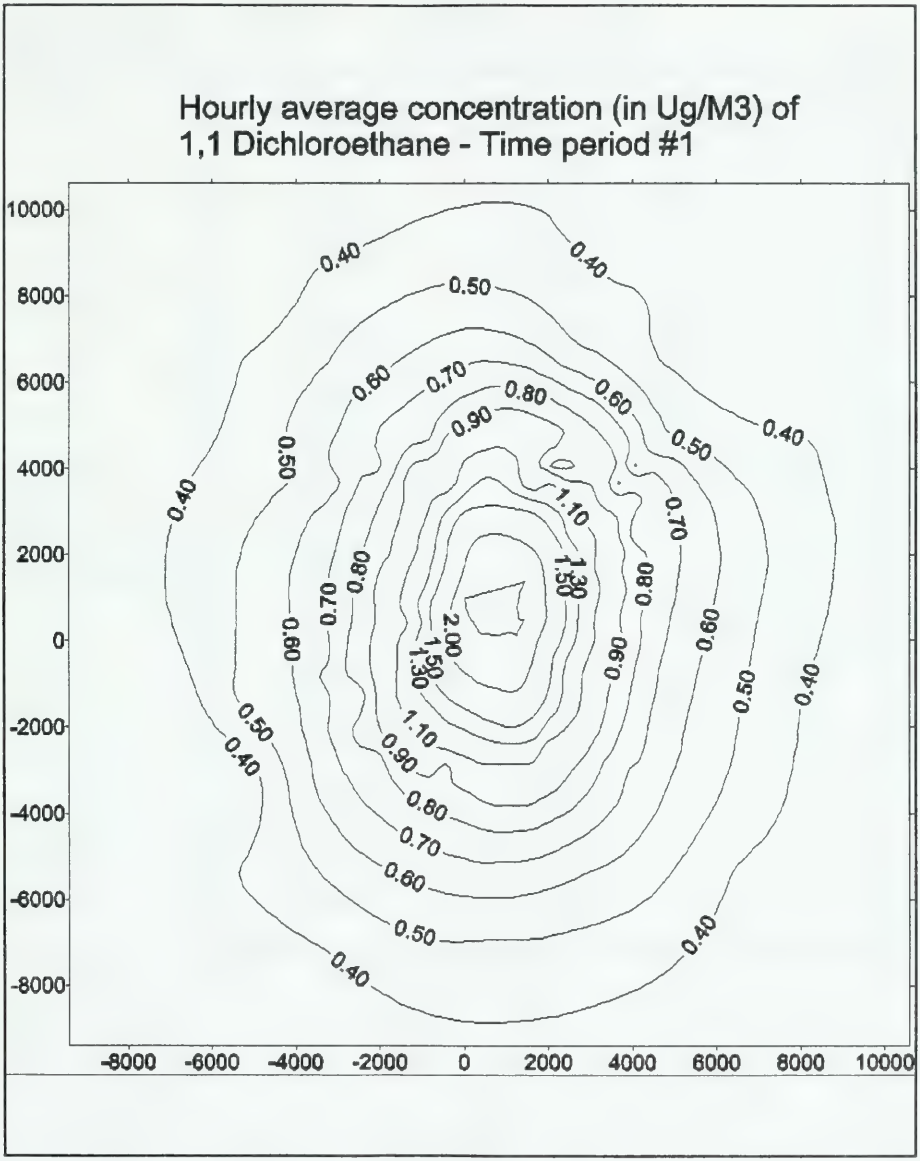


Figure B-II-6 Modelled maximum 1 hour average concentrations for 1,1-dichloroethane (Period 1).

Table B-I-1 Annual and Cumulative Waste Deposition

End of Year	Annual Waste Quantity (tonnes)	Waste in Place (tonnes)
1	316800	316800
2	305800	622600
3	298000	921400
4	290900	1212300
5	282000	1494000
6	289600	1783900
7	296400	2080300
8	303400	2383700
9	310400	2694100
10	317400	3011500
11	324400	3335900
12	331300	3667200
13	338300	4005500
14	345200	4350700
15	352400	4703100
16	359600	5062700
17	365200	5427900
18	370900	5798800
19	376400	6175200
20	381900	6557100
Total	6557100	

Table B-I-2: LANDFILL GAS GENERATION, GAS FLOW TO FLARE AND AND SURFACE EMISSIONS

END OF YEAR	LANDFILL STATUS	CH4 GENERATION (ft3/yr)	LFG GENERATION (ft3/yr)	COVER EMISSIONS (scfm/yr)	FLARE INLET (scfm/yr)
4	OPEN	2.00E+08	4.00E+08	4.00E+08	0
8	OPEN	3.67E+08	7.34E+08	3.67E+08	3.67E+08
12	OPEN	5.26E+08	1.05E+09	5.26E+08	5.26E+08
16	OPEN	6.81E+08	1.36E+09	6.81E+08	6.81E+08
20	OPEN	8.29E+08	1.66E+09	8.29E+08	8.29E+08
25	CLOSED	6.78E+08	1.36E+09	4.07E+08	9.49E+08
30	CLOSED	5.56E+08	1.11E+09	3.34E+08	7.78E+08
40	CLOSED	3.72E+08	7.44E+08	2.23E+08	5.21E+08

Assumes no LFG collection system until end of year 4

Assumes 50% collection efficiency from year 5 to end of year 20

Assumes 70% collection efficiency from year 21 to year 40

Table B-I-3: CONTAMINANT CONCENTRATION AND RANGES USED IN THE MODELLING

POLLUTANT	LANDFILL GAS CONCENTRATION						HIGH/LOW CONC.		RANGES AND VALUES USED		
	Britannia (ppm)	USEPA (ppm)	Keele Valley (ppm)	Beare Road (ppm)	Essex No. 3 (ppm)	Eastview (ppm)	Low Value (ppm)	High Value (ppm)	Value Used (ppm)	5th Percentile (ppm)	95th Percentile (ppm)
VINYL CHLORIDE	1.9	7.37	7.8		5.7	5.1	1.9	7.8	2.53	0.2	33
1,1-DICHLOROETHANE	9.61	2.07			0.06	1.64	0.06	9.61	9.61	0.75	120
BROMODICHLOROMETHANE		2.06				<0.01	0	2.06	2.06	0.16	27
1,1-DICHLOROETHYLENE	0.75	2.07			0.1	0.49	0.1	2.07	0.75	0.058	9.7
1,1,2,2-TETRACHLOROETHANE		0.2				0.05	0.05	0.2	0.2	0.016	2.6
METHYLENE CHLORIDE	26.4	14.3	56.35	1.6		9.97	1.6	56.35	26.4	2.0	340
TRICHLOROETHYLENE	2	2.08	2.2	1.7	2.7	1.05	1.05	2.7	2	0.016	26
1,2-DICHLOROETHANE	0.11	0.79			0.05	0.03	0.03	0.79	0.11	0.0085	1.4
1,1,2-TRICHLOROETHANE		0.1				0.13	0.1	0.13	0.1	0.0078	1.3
CHLOROFORM	0.039	0.27			0.1	<0.02	0	0.27	0.039	0.003	0.5
CHLOROMETHANE	0.95	1.14				0.08	0.08	1.14	0.95	0.074	12
CARBON TETRACHLORIDE	0.0079	0				<0.02	0	0.0079	0.0079	0.00061	0.1
CHLOROETHANE		1.17				1.15	1.15	1.17	1.17	0.091	15
METHYL MERCAPTAN		10.43					10.43	10.43	10.43	0.81	130
ETHYL MERCAPTAN				5.4			5.4	5.4	5.4	0.42	70
CARBONYL SULPHIDE		24		8.4			8.4	24	8.4	0.65	1100
OCTANE				7.7			7.7	7.7	7.7	0.6	99
BUTAN-2-OL	39.6						39.6	39.6	39.6	3.1	510
DIMETHYL SULPHIDE	3.2	76.16					3.2	76.16	3.2	0.25	41
HYDROGEN SULPHIDE	0.6	36.51	58				0.6	58	0.6	0.047	7.7
BENZENE	0.68	0.37	0.89		0.45		0.37	0.89	0.68	0.053	8.8

values used are based on Britannia data, supplemented by Other Ontario data and then USEPA
Medians used where available
Average used where medians are not compiled
Single non-detected values not used in deriving the GSD
Ranges not used or recorded
Geo. mean and GSD only use data set with more than 2 points

Table B-I-4a: YEAR 1-4 DEPOSIT SURFACE NMOC EMISSIONS

CONTAMINANT	COVER EMISSION FOR YEARS (g/s total covered areas)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	2.4E-03	1.0E-03	8.5E-04	7.4E-04	6.3E-04	3.1E-04	2.5E-04	1.7E-04
1,1-DICHLOROETHANE	1.4E-02	6.1E-03	5.1E-03	4.5E-03	3.8E-03	1.9E-03	1.5E-03	1.0E-03
BROMODICHLOROMETHANE	5.1E-03	2.2E-03	1.8E-03	1.6E-03	1.3E-03	6.6E-04	5.4E-04	3.6E-04
1,1-DICHLOROETHYLENE	1.1E-03	4.7E-04	3.9E-04	3.4E-04	2.9E-04	1.4E-04	1.2E-04	7.8E-05
1,1,2,2-TETRACHLOROETHANE	5.1E-04	2.2E-04	1.8E-04	1.6E-04	1.3E-04	6.6E-05	5.4E-05	3.6E-05
METHYLENE CHLORIDE	3.4E-02	1.4E-02	1.2E-02	1.1E-02	9.0E-03	4.4E-03	3.6E-03	2.4E-03
TRICHLOROETHYLENE	4.0E-03	1.7E-03	1.4E-03	1.2E-03	1.0E-03	5.1E-04	4.2E-04	2.8E-04
1,2-DICHLOROETHANE	1.6E-04	7.0E-05	5.8E-05	5.1E-05	4.3E-05	2.1E-05	1.7E-05	1.2E-05
1,1,2-TRICHLOROETHANE	2.0E-04	8.6E-05	7.1E-05	6.2E-05	5.3E-05	2.6E-05	2.1E-05	1.4E-05
CHLOROFORM	7.0E-05	3.0E-05	2.5E-05	2.2E-05	1.9E-05	9.1E-06	7.5E-06	5.0E-06
CHLOROMETHANE	7.3E-04	3.1E-04	2.6E-04	2.3E-04	1.9E-04	9.5E-05	7.8E-05	5.2E-05
CARBON TETRACHLORIDE	1.8E-05	7.8E-06	6.5E-06	5.7E-06	4.9E-06	2.4E-06	2.0E-06	1.3E-06
CHLOROETHANE	1.2E-03	4.9E-04	4.1E-04	3.6E-04	3.0E-04	1.5E-04	1.2E-04	8.2E-05
METHYL MERCAPTAN	7.6E-03	3.2E-03	2.7E-03	2.3E-03	2.0E-03	9.8E-04	8.0E-04	5.4E-04
ETHYL MERCAPTAN	5.1E-03	2.2E-03	1.8E-03	1.6E-03	1.3E-03	6.6E-04	5.4E-04	3.6E-04
CARBONYL SULPHIDE	7.6E-03	3.3E-03	2.7E-03	2.4E-03	2.0E-03	9.9E-04	8.1E-04	5.4E-04
OCTANE	1.3E-02	5.7E-03	4.7E-03	4.1E-03	3.5E-03	1.7E-03	1.4E-03	9.4E-04
BUTAN-2-OL	4.4E-02	1.9E-02	1.6E-02	1.4E-02	1.2E-02	5.7E-03	4.7E-03	3.2E-03
DIMETHYL SULPHIDE	3.0E-03	1.3E-03	1.1E-03	9.3E-04	7.9E-04	3.9E-04	3.2E-04	2.1E-04
HYDROGEN SULPHIDE	3.1E-04	1.3E-04	1.1E-04	9.6E-05	8.1E-05	4.0E-05	3.3E-05	2.2E-05
BENZENE	8.0E-04	3.4E-04	2.8E-04	2.5E-04	2.1E-04	1.0E-04	8.5E-05	5.7E-05

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in generation
Assumes no LFG collection system from years 1-4

Table B-I-4b: YEARS 1-4 DEPOSITS UNIT AREA NMOC EMISSION RATES

CONTAMINANT	COVER EMISSION FOR YEARS (g/s/m2 total cover areas)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	1.6E-08	7.0E-09	5.8E-09	5.1E-09	4.3E-09	2.1E-09	1.7E-09	1.2E-09
1,1-DICHLOROETHANE	9.8E-08	4.2E-08	3.5E-08	3.0E-08	2.6E-08	1.3E-08	1.0E-08	7.0E-09
BROMODICHLOROMETHANE	3.5E-08	1.5E-08	1.2E-08	1.1E-08	9.2E-09	4.5E-09	3.7E-09	2.5E-09
1,1-DICHLOROETHYLENE	7.5E-09	3.2E-09	2.7E-09	2.3E-09	2.0E-09	9.7E-10	8.0E-10	5.3E-10
1,1,2,2-TETRACHLOROETHANE	3.5E-09	1.5E-09	1.2E-09	1.1E-09	9.2E-10	4.5E-10	3.7E-10	2.5E-10
METHYLENE CHLORIDE	2.3E-07	9.9E-08	8.2E-08	7.2E-08	6.1E-08	3.0E-08	2.5E-08	1.6E-08
TRICHLOROETHYLENE	2.7E-08	1.2E-08	9.6E-09	8.4E-09	7.1E-09	3.5E-09	2.9E-09	1.9E-09
1,2-DICHLOROETHANE	1.1E-09	4.8E-10	4.0E-10	3.5E-10	3.0E-10	1.5E-10	1.2E-10	8.0E-11
1,1,2-TRICHLOROETHANE	1.4E-09	5.9E-10	4.9E-10	4.3E-10	3.6E-10	1.8E-10	1.5E-10	9.8E-11
CHLOROFORM	4.8E-10	2.0E-10	1.7E-10	1.5E-10	1.3E-10	6.2E-11	5.1E-11	3.4E-11
CHLOROMETHANE	5.0E-09	2.1E-09	1.8E-09	1.5E-09	1.3E-09	6.5E-10	5.3E-10	3.6E-10
CARBON TETRACHLORIDE	1.3E-10	5.4E-11	4.5E-11	3.9E-11	3.3E-11	1.6E-11	1.3E-11	8.9E-12
CHLOROETHANE	7.9E-09	3.4E-09	2.8E-09	2.4E-09	2.1E-09	1.0E-09	8.3E-10	5.6E-10
METHYL MERCAPTAN	5.2E-08	2.2E-08	1.8E-08	1.6E-08	1.4E-08	6.7E-09	5.5E-09	3.7E-09
ETHYL MERCAPTAN	3.5E-08	1.5E-08	1.2E-08	1.1E-08	9.1E-09	4.5E-09	3.7E-09	2.5E-09
CARBONYL SULPHIDE	5.2E-08	2.2E-08	1.8E-08	1.6E-08	1.4E-08	6.7E-09	5.5E-09	3.7E-09
OCTANE	9.1E-08	3.9E-08	3.2E-08	2.8E-08	2.4E-08	1.2E-08	9.6E-09	6.5E-09
BUTAN-2-OL	3.0E-07	1.3E-07	1.1E-07	9.4E-08	8.0E-08	3.9E-08	3.2E-08	2.2E-08
DIMETHYL SULPHIDE	2.1E-08	8.7E-09	7.3E-09	6.3E-09	5.4E-09	2.7E-09	2.2E-09	1.5E-09
HYDROGEN SULPHIDE	2.1E-09	9.0E-10	7.5E-10	6.5E-10	5.6E-10	2.7E-10	2.2E-10	1.5E-10
BENZENE	5.5E-09	2.3E-09	1.9E-09	1.7E-09	1.4E-09	7.1E-10	5.8E-10	3.9E-10

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in generation

Table B-I-4c: YEARS 5-8 DEPOSIT SURFACE NMOC EMISSIONS

CONTAMINANT	COVER EMISSION FOR YEARS (g/s total cover areas)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	0.0	1.2E-03	9.9E-04	8.4E-04	7.2E-04	3.5E-04	2.9E-04	1.9E-04
1,1-DICHLOROETHANE	0.0	7.0E-03	5.9E-03	5.1E-03	4.3E-03	2.1E-03	1.7E-03	1.2E-03
BROMODICHLOROMETHANE	0.0	2.5E-03	2.1E-03	1.8E-03	1.5E-03	7.5E-04	6.2E-04	4.1E-04
1,1-DICHLOROETHYLENE	0.0	5.3E-04	4.5E-04	3.9E-04	3.3E-04	1.6E-04	1.3E-04	8.9E-05
1,1,2,2-TETRACHLOROETHANE	0.0	2.5E-04	2.1E-04	1.8E-04	1.5E-04	7.5E-05	6.1E-05	4.1E-05
METHYLENE CHLORIDE	0.0	1.6E-02	1.4E-02	1.2E-02	1.0E-02	5.0E-03	4.1E-03	2.7E-03
TRICHLOROETHYLENE	0.0	1.9E-03	1.6E-03	1.4E-03	1.2E-03	5.8E-04	4.8E-04	3.2E-04
1,2-DICHLOROETHANE	0.0	8.0E-05	6.8E-05	5.8E-05	4.9E-05	2.4E-05	2.0E-05	1.3E-05
1,1,2-TRICHLOROETHANE	0.0	9.8E-05	8.3E-05	7.1E-05	6.0E-05	3.0E-05	2.4E-05	1.6E-05
CHLOROFORM	0.0	3.4E-05	2.9E-05	2.5E-05	2.1E-05	1.0E-05	8.5E-06	5.7E-06
CHLOROMETHANE	0.0	3.6E-04	3.0E-04	2.6E-04	2.2E-04	1.1E-04	8.8E-05	5.9E-05
CARBON TETRACHLORIDE	0.0	8.9E-06	7.6E-06	6.5E-06	5.5E-06	2.7E-06	2.2E-06	1.5E-06
CHLOROETHANE	0.0	5.6E-04	4.8E-04	4.0E-04	3.5E-04	1.7E-04	1.4E-04	9.3E-05
METHYL MERCAPTAN	0.0	3.7E-03	3.1E-03	2.7E-03	2.3E-03	1.1E-03	9.1E-04	6.1E-04
ETHYL MERCAPTAN	0.0	2.5E-03	2.1E-03	1.8E-03	1.5E-03	7.5E-04	6.1E-04	4.1E-04
CARBONYL SULPHIDE	0.0	3.7E-03	3.1E-03	2.7E-03	2.3E-03	1.1E-03	9.2E-04	6.2E-04
OCTANE	0.0	6.4E-03	5.5E-03	4.7E-03	4.0E-03	2.0E-03	1.6E-03	1.1E-03
BUTAN-2-OL	0.0	2.1E-02	1.8E-02	1.6E-02	1.3E-02	6.5E-03	5.3E-03	3.6E-03
DIMETHYL SULPHIDE	0.0	1.5E-03	1.2E-03	1.1E-03	9.0E-04	4.4E-04	3.6E-04	2.4E-04
HYDROGEN SULPHIDE	0.0	1.5E-04	1.3E-04	1.1E-04	9.3E-05	4.5E-05	3.7E-05	2.5E-05
BENZENE	0.0	3.9E-04	3.3E-04	2.8E-04	2.4E-04	1.2E-04	9.7E-05	6.5E-05

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in generation

Table B-I-4d: YEAR 5-8 DEPOSITS UNIT AREA NMOC EMISSION RATES

CONTAMINANT	COVER EMISSION FOR YEARS (g/s/m2 total cover areas)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	0.0	1.3E-08	1.1E-08	9.3E-09	7.9E-09	3.9E-09	3.2E-09	2.1E-09
1,1-DICHLOROETHANE	0.0	7.7E-08	6.6E-08	5.6E-08	4.8E-08	2.3E-08	1.9E-08	1.3E-08
BROMODICHLOROMETHANE	0.0	2.7E-08	2.3E-08	2.0E-08	1.7E-08	8.3E-09	6.8E-09	4.6E-09
1,1-DICHLOROETHYLENE	0.0	5.9E-09	5.0E-09	4.3E-09	3.6E-09	1.8E-09	1.5E-09	9.8E-10
1,1,2,2-TETRACHLOROETHANE	0.0	2.7E-09	2.3E-09	2.0E-09	1.7E-09	8.3E-10	6.8E-10	4.5E-10
METHYLENE CHLORIDE	0.0	1.8E-07	1.5E-07	1.3E-07	1.1E-07	5.5E-08	4.5E-08	3.0E-08
TRICHLOROETHYLENE	0.0	2.1E-08	1.8E-08	1.5E-08	1.3E-08	6.4E-09	5.3E-09	3.5E-09
1,2-DICHLOROETHANE	0.0	8.8E-10	7.5E-10	6.4E-10	5.5E-10	2.7E-10	2.2E-10	1.5E-10
1,1,2-TRICHLOROETHANE	0.0	1.1E-09	9.2E-10	7.8E-10	6.7E-10	3.3E-10	2.7E-10	1.8E-10
CHLOROFORM	0.0	3.8E-10	3.2E-10	2.7E-10	2.3E-10	1.1E-10	9.3E-11	6.3E-11
CHLOROMETHANE	0.0	3.9E-09	3.3E-09	2.8E-09	2.4E-09	1.2E-09	9.8E-10	6.5E-10
CARBON TETRACHLORIDE	0.0	9.8E-11	8.4E-11	7.1E-11	6.1E-11	3.0E-11	2.4E-11	1.6E-11
CHLOROETHANE	0.0	6.2E-09	5.2E-09	4.5E-09	3.8E-09	1.9E-09	1.5E-09	1.0E-09
METHYL MERCAPTAN	0.0	4.0E-08	3.5E-08	2.9E-08	2.5E-08	1.2E-08	1.0E-08	6.8E-09
ETHYL MERCAPTAN	0.0	2.7E-08	2.3E-08	2.0E-08	1.7E-08	8.2E-09	6.7E-09	4.5E-09
CARBONYL SULPHIDE	0.0	4.1E-08	3.5E-08	3.0E-08	2.5E-08	1.2E-08	1.0E-08	6.8E-09
OCTANE	0.0	7.1E-08	6.1E-08	5.2E-08	4.4E-08	2.2E-08	1.8E-08	1.2E-08
BUTAN-2-OL	0.0	2.4E-07	2.0E-07	1.7E-07	1.5E-07	7.2E-08	5.9E-08	4.0E-08
DIMETHYL SULPHIDE	0.0	1.6E-08	1.4E-08	1.2E-08	9.9E-09	4.9E-09	4.0E-09	2.7E-09
HYDROGEN SULPHIDE	0.0	1.6E-09	1.4E-09	1.2E-09	1.0E-09	5.0E-10	4.1E-10	2.8E-10
BENZENE	0.0	4.3E-09	3.7E-09	3.1E-09	2.7E-09	1.3E-09	1.1E-09	7.2E-10

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in LFG generation

Table B-I-4e: YEAR 9-20 SURFACE NMOC EMISSIONS

CONTAMINANT	COVER EMISSION FOR YEARS (g/s total cover areas)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	0.0	0.0	1.3E-03	2.5E-03	3.6E-03	1.8E-03	1.4E-03	9.6E-04
1,1-DICHLOROETHANE	0.0	0.0	7.6E-03	1.5E-02	2.2E-02	1.1E-02	8.7E-03	5.8E-03
BROMODICHLOROMETHANE	0.0	0.0	2.7E-03	5.3E-03	7.6E-03	3.8E-03	3.1E-03	2.1E-03
1,1-DICHLOROETHYLENE	0.0	0.0	5.8E-04	1.1E-03	1.6E-03	8.1E-04	6.6E-04	4.4E-04
1,1,2,2-TETRACHLOROETHANE	0.0	0.0	2.7E-04	5.2E-04	7.6E-04	3.7E-04	3.1E-04	2.0E-04
METHYLENE CHLORIDE	0.0	0.0	1.8E-02	3.5E-02	5.1E-02	2.5E-02	2.0E-02	1.4E-02
TRICHLOROETHYLENE	0.0	0.0	2.1E-03	4.1E-03	5.9E-03	2.9E-03	2.4E-03	1.6E-03
1,2-DICHLOROETHANE	0.0	0.0	8.8E-05	1.7E-04	2.5E-04	1.2E-04	9.9E-05	6.6E-05
1,1,2-TRICHLOROETHANE	0.0	0.0	1.1E-04	2.1E-04	3.0E-04	1.5E-04	1.2E-04	8.1E-05
CHLOROFORM	0.0	0.0	3.7E-05	7.2E-05	1.1E-04	5.2E-05	4.2E-05	2.8E-05
CHLOROMETHANE	0.0	0.0	3.9E-04	7.5E-04	1.1E-03	5.4E-04	4.4E-04	3.0E-04
CARBON TETRACHLORIDE	0.0	0.0	9.8E-06	1.9E-05	2.8E-05	1.4E-05	1.1E-05	7.4E-06
CHLOROETHANE	0.0	0.0	6.1E-04	1.2E-03	1.7E-03	8.5E-04	6.9E-04	4.6E-04
METHYL MERCAPTAN	0.0	0.0	4.0E-03	7.8E-03	1.1E-02	5.6E-03	4.6E-03	3.1E-03
ETHYL MERCAPTAN	0.0	0.0	2.7E-03	5.2E-03	7.6E-03	3.7E-03	3.0E-03	2.0E-03
CARBONYL SULPHIDE	0.0	0.0	4.0E-03	7.9E-03	1.1E-02	5.6E-03	4.6E-03	3.1E-03
OCTANE	0.0	0.0	7.1E-03	1.4E-02	2.0E-02	9.8E-03	8.0E-03	5.4E-03
BUTAN-2-OL	0.0	0.0	2.4E-02	4.6E-02	6.6E-02	3.3E-02	2.7E-02	1.8E-02
DIMETHYL SULPHIDE	0.0	0.0	1.6E-03	3.1E-03	4.5E-03	2.2E-03	1.8E-03	1.2E-03
HYDROGEN SULPHIDE	0.0	0.0	1.6E-04	3.2E-04	4.6E-04	2.3E-04	1.9E-04	1.2E-04
BENZENE	0.0	0.0	4.3E-04	8.3E-04	1.2E-03	5.9E-04	4.8E-04	3.2E-04

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in generation

Table B-I-4f: YEARS 9-20 DEPOSITS UNIT AREA NMOC EMISSION RATE

CONTAMINANT	COVER EMISSION FOR YEARS (g/s/m2 total cover areas)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	0.0	0.0	7.6E-09	1.5E-08	2.1E-08	1.1E-08	8.6E-09	5.8E-09
1,1-DICHLOROETHANE	0.0	0.0	4.6E-08	8.9E-08	1.3E-07	6.3E-08	5.2E-08	3.5E-08
BROMODICHLOROMETHANE	0.0	0.0	1.6E-08	3.1E-08	4.6E-08	2.2E-08	1.8E-08	1.2E-08
1,1-DICHLOROETHYLENE	0.0	0.0	3.5E-09	6.8E-09	9.9E-09	4.8E-09	4.0E-09	2.7E-09
1,1,2,2-TETRACHLOROETHANE	0.0	0.0	1.6E-09	3.1E-09	4.6E-09	2.2E-09	1.8E-09	1.2E-09
METHYLENE CHLORIDE	0.0	0.0	1.1E-07	2.1E-07	3.0E-07	1.5E-07	1.2E-07	8.2E-08
TRICHLOROETHYLENE	0.0	0.0	1.3E-08	2.4E-08	3.5E-08	1.7E-08	1.4E-08	9.6E-09
1,2-DICHLOROETHANE	0.0	0.0	5.2E-10	1.0E-09	1.5E-09	7.2E-10	5.9E-10	4.0E-10
1,1,2-TRICHLOROETHANE	0.0	0.0	6.4E-10	1.2E-09	1.8E-09	8.9E-10	7.2E-10	4.9E-10
CHLOROFORM	0.0	0.0	2.2E-10	4.3E-10	6.3E-10	3.1E-10	2.5E-10	1.7E-10
CHLOROMETHANE	0.0	0.0	2.3E-09	4.5E-09	6.6E-09	3.2E-09	2.6E-09	1.8E-09
CARBON TETRACHLORIDE	0.0	0.0	5.9E-11	1.1E-10	1.6E-10	8.1E-11	6.6E-11	4.4E-11
CHLOROETHANE	0.0	0.0	3.7E-09	7.1E-09	1.0E-08	5.1E-09	4.1E-09	2.8E-09
METHYL MERCAPTAN	0.0	0.0	2.4E-08	4.7E-08	6.8E-08	3.3E-08	2.7E-08	1.8E-08
ETHYL MERCAPTAN	0.0	0.0	1.6E-08	3.1E-08	4.5E-08	2.2E-08	1.8E-08	1.2E-08
CARBONYL SULPHIDE	0.0	0.0	2.4E-08	4.7E-08	6.8E-08	3.4E-08	2.7E-08	1.8E-08
OCTANE	0.0	0.0	4.2E-08	8.2E-08	1.2E-07	5.8E-08	4.8E-08	3.2E-08
BUTAN-2-OL	0.0	0.0	1.4E-07	2.7E-07	4.0E-07	1.9E-07	1.6E-07	1.1E-07
DIMETHYL SULPHIDE	0.0	0.0	9.5E-09	1.9E-08	2.7E-08	1.3E-08	1.1E-08	7.2E-09
HYDROGEN SULPHIDE	0.0	0.0	9.8E-10	1.9E-09	2.8E-09	1.4E-09	1.1E-09	7.4E-10
BENZENE	0.0	0.0	2.6E-09	4.9E-09	7.2E-09	3.5E-09	2.9E-09	1.9E-09

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in LFG generation

Table B-I-5 ENCLOSED FLARE STACK PARAMETERS

Year	LFG Flow into the Flare* (m^3/s @ 60°F)	Total Combustion Gas Exhaust Rate (actual m^3/s @ 1800 °F)	Flare* Exhaust Gas Exit Velocity (m/s)
8	0.17	7.0	2.7
12	0.24	10.0	3.9
16	0.31	12.9	4.9
20	0.37	15.7	6.0
21	0.50	21.0	8.1
25	0.43	18.0	6.9
30	0.35	14.7	5.7
40	0.23	9.8	3.8

* Flare is 6 ft in diameter and 34 feet tall.
Flame front is 5 ft from the base at a temperature of 2200 °F.
Combustion Exit temperature is 1800 °F for all years.

Table B-I-6: ENCLOSED FLARE NMOC EMISSION RATES

CONTAMINANT	FLARE EMISSION FOR YEARS (g/s all flares)							
	4	8	12	16	20	25	30	40
VINYL CHLORIDE	0.0	4.4E-05	6.3E-05	8.1E-05	9.9E-05	1.1E-04	9.3E-05	6.2E-05
1,1-DICHLOROETHANE	0.0	2.6E-04	3.8E-04	4.9E-04	5.9E-04	6.8E-04	5.6E-04	3.7E-04
BROMODICHLOROMETHANE	0.0	9.3E-05	1.3E-04	1.7E-04	2.1E-04	2.4E-04	2.0E-04	1.3E-04
1,1-DICHLOROETHYLENE	0.0	2.0E-05	2.9E-05	3.7E-05	4.5E-05	5.2E-05	4.3E-05	2.8E-05
1,1,2,2-TETRACHLOROETHANE	0.0	9.3E-06	1.3E-05	1.7E-05	2.1E-05	2.4E-05	2.0E-05	1.3E-05
METHYLENE CHLORIDE	0.0	6.2E-04	8.9E-04	1.1E-03	1.4E-03	1.6E-03	1.3E-03	8.8E-04
TRICHLOROETHYLENE	0.0	7.2E-05	1.0E-04	1.3E-04	1.6E-04	1.9E-04	1.5E-04	1.0E-04
1,2-DICHLOROETHANE	0.0	3.0E-06	4.3E-06	5.6E-06	6.8E-06	7.8E-06	6.4E-06	4.3E-06
1,1,2-TRICHLOROETHANE	0.0	3.7E-06	5.3E-06	6.8E-06	8.3E-06	9.5E-06	7.8E-06	5.2E-06
CHLOROFORM	0.0	1.3E-06	1.8E-06	2.4E-06	2.9E-06	3.3E-06	2.7E-06	1.8E-06
CHLOROMETHANE	0.0	1.3E-05	1.9E-05	2.5E-05	3.0E-05	3.5E-05	2.8E-05	1.9E-05
CARBON TETRACHLORIDE	0.0	3.4E-07	4.8E-07	6.2E-07	7.6E-07	8.7E-07	7.1E-07	4.8E-07
CHLOROETHANE	0.0	2.1E-05	3.0E-05	3.9E-05	4.7E-05	5.4E-05	4.5E-05	3.0E-05
METHYL MERCAPTAN	0.0	1.4E-04	2.0E-04	2.6E-04	3.1E-04	3.6E-04	2.9E-04	2.0E-04
ETHYL MERCAPTAN	0.0	9.2E-05	1.3E-04	1.7E-04	2.1E-04	2.4E-04	2.0E-04	1.3E-04
CARBONYL SULPHIDE	0.0	1.4E-04	2.0E-04	2.6E-04	3.1E-04	3.6E-04	3.0E-04	2.0E-04
OCTANE	0.0	2.4E-04	3.5E-04	4.5E-04	5.5E-04	6.3E-04	5.1E-04	3.4E-04
BUTAN-2-OL	0.0	8.1E-04	1.2E-03	1.5E-03	1.8E-03	2.1E-03	1.7E-03	1.1E-03
DIMETHYL SULPHIDE	0.0	5.5E-05	7.8E-05	1.0E-04	1.2E-04	1.4E-04	1.2E-04	7.8E-05
HYDROGEN SULPHIDE	0.0	5.6E-06	8.1E-06	1.0E-05	1.3E-05	1.5E-05	1.2E-05	8.0E-06
BENZENE	0.0	1.5E-05	2.1E-05	2.7E-05	3.3E-05	3.8E-05	3.1E-05	2.1E-05

Assumes 50% LFG collection efficiency for years 0-20
Assumes 70 collection control efficiency for years 20 and up
Assumes no lag in generation
Assumes 98% detruction efficiency in the flare

APPENDIX B-II Conditions And Assumptions Used In The Deterministic And Probabilistic Multimedia Exposure Assessment Model.

Appendix B-II-A. Distributions and Point Values for Exposure Variables.

Parameter	Distribution	Mean or Likeliest	Standard Deviation	Minimum	Maximum	Reference
Body Weight per Age Group (kg)						
0 - 6m	LN	6.28	1.32	3.32	11.7	IWA, 1994
7m - 4y	LN	13.0	2.99	6.28	25.6	IWA, 1994
5y - 12y	LN	26.8	8.64	9.65	67.4	IWA, 1994
13y - 20y	LN	50.0	9.87	26.8	89.9	IWA, 1994
21+y	LN	61.5	11.1	34.8	105.0	IWA, 1994
Inhalation Rate per Age Group (m^3/d)						
0 - 6m	T	1.21	-	0.36	3.01	IWA, 1994
7m - 4y	T	11.0	-	10.8	11.2	IWA, 1994
5y - 12y	T	15.9	-	11.7	20.7	IWA, 1994
13y - 20y	T	26.1	-	15.6	29.2	IWA, 1994
21+y	T	21.4	-	7.69	32.0	IWA, 1994
Surface Area per Age Group (cm^2)						
7m - 4y	U	-	-	1310	2140	IWA, 1994
5y - 12y	N	2380	320	1390	3370	IWA, 1994
13y - 20y	N	3630	300	2700	4560	IWA, 1994
21+y	N	3400	142	2960	3400	IWA, 1994
Soil Loading Factor (mg/cm^2-day)						
	LN	0.52	0.9	-∝	+∝	Finley et al., 1994
Soil Ingestion per Age Group (mg/day)						

7m - 4y	LN	91	126	2.2	1300	IWA, 1994
5y - 12y	LN	91	126	2.2	1300	IWA, 1994
13y - 20y	U	-	-	1	250	IWA, 1994
21+y	U	-	-	1	100	IWA, 1994
Leafy Crop Ingestion Rate per Age Group (kg/day)						
0 - 6m	LN	1.54E-2	1.39E-2	1.05E-3	1.24E-1	IWA, 1994
7m - 4y	LN	3.72E-2	3.35E-2	2.55E-3	2.99E-1	IWA, 1994
5y - 12y	LN	4.75E-2	4.28E-2	3.26E-3	3.82E-1	IWA, 1994
13y - 20y	LN	4.56E-2	3.06E-2	5.75E-3	2.49E-1	IWA, 1994
21+y	LN	4.28E-2	2.87E-2	5.41E-3	2.34E-1	IWA, 1994
Fruit Ingestion Rate per Age Group (kg/day)						
0 - 6m	LN	1.12E-1	1.01E-1	7.0E-2	9.02E-1	IWA, 1994
7m - 4y	LN	2.62E-1	2.36E-1	1.8E-2	2.1E+0	IWA, 1994
5y - 12y	LN	3.05E-1	2.75E-1	2.09E-2	2.46+0	IWA, 1994
13y - 20y	LN	2.87E-1	1.92E-1	3.64E-2	1.56+0	IWA, 1994
21+y	LN	2.37E-1	1.59E-1	2.99E-2	1.29E+0	IWA, 1994
Vegetable Ingestion Rate per Age Group (kg/day)						
0 - 6m	LN	4.2E-2	3.78E-2	3.0E-3	3.37E-1	IWA, 1994
7m - 4y	LN	2.62E-1	2.36E-1	1.8E-2	2.1+0	IWA, 1994
5y - 12y	LN	7.1E-2	6.39E-2	4.88E-3	5.7E-1	IWA, 1994
13y - 20y	LN	9.5E-2	6.37E-2	1.2E-2	5.19E-1	IWA, 1994
21+y	LN	1.14E-1	7.64E-2	1.44E-2	6.22E-1	IWA, 1994
Root Crop Ingestion Rate per Age Group (kg/day)						

7m - 4y	LN	6.1E-2	5.49E-2	4.19E-3	4.9E-1	IWA, 1994
5y - 12y	LN	9.9E-2	8.91E-2	6.81E-3	7.95E-1	IWA, 1994
13y - 20y	LN	7.4E-2	4.96E-2	9.36E-3	4.04E-1	IWA, 1994
21+y	LN	7.6E-2	5.09E-2	9.62E-3	4.14E-1	IWA, 1994
Water Ingestion Rate per Age Group (L/day)						
0 - 6m	U			0.4	0.8	IWA, 1994
7m - 4y	LN	0.83	0.77	0.17	2.22	IWA, 1994
5y - 12y	LN	1	0.68	0.12	5.56	IWA, 1994
13y - 20y	LN	1	0.68	0.12	5.56	IWA, 1994
21+y	LN	1.51	0.83	0.27	6.43	IWA, 1994
Indoor / Outdoor Ratio						
	T	0.75	-	0.2	1.0	Hawkins, 1991
LN: log-normal distribution; U: uniform distribution; T: triangular distribution; N: normal distribution						

Appendix B-II-B. Statistical Distributions for Parameters Used in the Probailistic Multimedia Model.

Parameter	Distribution	Mean	Standard Deviation	Minimum	Maximum	Reference
Soil Bulk Density (kg/ m^3)	U	1,401		1,000	1,800	Copeland et al., 1994
Deposition rate (m/day)	LN	6.37E-03	9.70E-03	- ∝	+ ∝	McKone and Daniels, 1991
Interception fraction (Leafy crops)	U	0.27		0.15	0.39	Copeland et al., 1994
Interception fraction (Vine crops)	U	0.05		0.03	0.07	Copeland et al., 1994
Crop Yield	U	3.2		2.0	4.4	Copeland et al., 1994
Growth Period (days)	U	67.5		45.0	90.0	Copeland et al., 1994
Senescence/ Weathering Period (days)	LN	10.00	3.00	2.80	34.00	Copeland et al., 1994

LN: log-normal distribution; U: uniform distribution; T: triangular distribution; N: normal distribution

APPENDIX B-II-C. Receptors characterisitcs - Comparison of Intake Rates*

Soil Ingestion		Inhalation rate		Water consumption	
g/kg-day (ratio}		m3/kg-day (ratio}		L/kg-day (ratio}	
6.00E-03	1.23E+00	1.90E-01	1.13E+00	1.30E-01	1.17E+00
4.86E-03		1.68E-01		1.11E-01	
7.00E-03	1.05E+00	8.46E-01	1.05E+00	6.38E-02	1.20E+00
6.64E-03		8.03E-01		5.33E-02	
3.40E-03	1.00E+00	5.93E-01	7.87E-01	3.73E-02	7.87E-01
3.40E-03		7.54E-01		4.74E-02	
5.00E-03	1.07E+00	5.22E-01	1.14E+00	2.00E-02	8.46E-01
4.66E-03		4.56E-01		2.36E-02	
1.63E-03	1.25E+00	3.48E-01	1.22E+00	2.46E-02	1.26E+00
1.30E-03		2.84E-01		1.96E-02	

* Data were determined by IWA (1994) based on EHD, 1993; EPA, 1985; Nutrition Canada, 1975; OMAF/OMOE, 1988 and other published material.

APPENDIX B-II-C. Receptors characterisitcs - Comparison of Intake Rates* (continued)

	Leafy plants consumption		Fruits consumption		Potatoes consumption		Veg-no pot consumpt.	
	g/kg-day (ratio)		g/kg-day (ratio)		g/kg-day (ratio)		g/kg-day (ratio)	
0-6m female	2.45E+00	1.15E+00	1.78E+01	1.14E+00	na		6.70E+00	1.15E+00
0-6m male	2.14E+00		1.56E+01		na		5.83E+00	
7m-4y female	2.86E+00	1.05E+00	2.02E+01	1.05E+00	4.69E+00	1.05E+00	3.77E+00	1.05E+00
7m-4y male	2.72E+00		1.91E+01		4.45E+00		3.58E+00	
5-12 female	1.77E+00	1.00E+00	1.14E+01	1.00E+00	3.69E+00	1.00E+00	2.65E+00	1.00E+00
5-12 male	1.77E+00		1.14E+01		3.69E+00		2.65E+00	
13-20 female	9.12E-01	9.09E-01	5.74E+00	1.17E+00	1.48E+00	4.62E-01	1.90E+00	9.81E-01
13-20 male	1.00E+00		4.90E+00		3.20E+00		1.94E+00	
21 + female	6.96E-01	1.06E+00	3.85E+00	1.26E+00	1.24E+00	6.92E-01	1.85E+00	1.08E+00
21 + male	6.56E-01		3.05E+00		1.79E+00		1.72E+00	

* Data were determined by IWA (1994) based on EHD, 1993; EPA, 1985; Nutrition Canada, 1975; OMAF/OMOE, 1988 and other published material.

APPENDIX B-III

Appendix B-III – A. Determination of Gas / Particulate Ratios for Air Emissions.

a) based on the fugacity model of MacKay, 1991

chemical	vapor pressure Pa	reference (a)	Kqa (b)	TSP.A/F (c)	gas / particulate ratio (d)	% in particulate
1,1,2,2-tetrachloroethane	440	MacKay et al., 1992	13636	1E+11	1E+06	7E-05
1,1,2-trichloroethane	2369	MacKay et al., 1992	2532.7	8E+11	8E+06	1E-05
trichloroethylene	7998	MacKay et al., 1992	750.19	3E+12	3E+07	4E-06
1,2-dichloroethane	8272	MacKay et al., 1992	725.34	3E+12	3E+07	4E-06
bromodichloromethane	8555	MacKay et al., 1992	701.34	3E+12	3E+07	4E-06
benzene	12690	MacKay et al., 1992	472.81	4E+12	4E+07	2E-06
1,1-dichloroethane	24440	MacKay et al., 1992	245.5	8E+12	8E+07	1E-06
methylene chloride	47656	MacKay et al., 1992	125.9	2E+13	2E+08	6E-07
ethyl mercaptan	58520	Verschueren, 1983	102.53	2E+13	2E+08	5E-07
methyl mercaptan	58520	Verschueren, 1983	102.53	2E+13	2E+08	5E-07
1,1-dichloroethylene	66188	MacKay et al., 1992	90.651	2E+13	2E+08	5E-07
chloroethane	133300	MacKay et al., 1992	45.011	4E+13	4E+08	2E-07
vinyl cholide	350000	MacKay et al., 1992	17.143	1E+14	1E+09	9E-08
hydrogen sulfide	2.03E+06	Verschueren, 1983	2.9608	7E+14	7E+09	1E-08

(a) Values extracted from Mackay et al., 1991 were the most recent reported at 20 C; those from Verschueren, 1983 were those at 20 C ; methylmercaptan assumed similar to ethylmercaptan

(b) Kqa: aerosol/air partition coefficient, where Kqa = 6E06/vapor pressure (MacKay, 1991)

(c) Magnitude of the partionning to air particulate, where TSP.A/F = 10E12 @/Kqa. An aerosol density, @, of 2000 kg/m3 was assumed (MacKay, 1991)

(d) Gas / particulate ratios were calculated as (TSP.A /F)/TSP by assuming a TSP of 100 ug/m3 (100000 ng/m3) (MacKay, 1991)

Appendix B-III – A. Determination of Gas / Particulate Ratios for Air Emissions (continued).

b) based on the methodology proposed by Travis and Hattemer-Frey, 1989

chemical	vapor pressure (a) mm Hg	F (b)
1,1,2,2-tetrachloroethane	3.31	3E-07
1,1,2-trichloroethane	17.81	6E-08
trichloroethylene	60.14	2E-08
1,2-dichloroethane	62.20	2E-08
bromodichloromethane	64.32	2E-08
benzene	95.41	1E-08
1,1-dichloroethane	183.76	5E-09
methylene chloride	358.32	3E-09
ethyl mercaptan	440.00	2E-09
methyl mercaptan	440.00	2E-09
1,1-dichloroethylene	497.65	2E-09
chloroethane	1002.26	1E-09
vinyl chloride	2631.58	4E-10
hydrogen sulfide	15236.84	7E-11

(a) See previous table

(b) Fraction sorbed to particulates; F = (K/vapor pressure) / (1 + K/vapor pressure) (Travis and Hattemer-Frey, 1989)

Appendix B-III – B. Screening of Carcinogenic Air Emission Components.

Results of the Screening Procedure for Carcinogens

Chemical	USEPA Class	Critical end-point	Unit Risk	Reference	RsC 10E-6
			(ug/m3)E-1		ug/m3
Chlorinated Organics					
vinyl chloride	A	liver tumors	8.40E-05	HEAST, 1992	0.0119
bromodichloromethane (d)	B2	kidney tumors	2.00E-05	IRIS, 1994	0.0500
1,1-dichloroethylene	C	kidney adenocarcinomas	5.00E-05	IRIS, 1994	0.0200
1,1,2,2-tetrachloroethane	C	Hepatocellular carcinomas	5.80E-05	IRIS, 1994	0.0172
1,1-dichloroethane	C	Several tumor types	5.70E-06	CalEPA, 1996	0.625
methylene chloride	B2	liver adenomas & carcinomas	4.70E-07	IRIS, 1994	2.1277
trichloroethylene	NA	NA	2.00E-06	CAPCOA, 1993	0.5000
1,2-dichloroethane	B2	Several tumor types	2.60E-05	IRIS, 1994	0.0385
1,1,2-trichloroethane	C	Hepatocellular carcinomas	1.60E-05	IRIS, 1994	0.0625
chloroethane (c)	C	kidney tumors	1.80E-06	IWA, 1994	0.5556
chloroform	B2	several tumor types	2.30E-05	IRIS, 1994	0.0435
chloromethane	C	kidney tumors	1.80E-06	HEAST, 1992	0.5556
carbon tetrachloride	B2	hepatocellular carcinomas	1.50E-05	IRIS, 1994	0.0667
Others					
benzene	A	Leukemia	8.30E-06	IRIS, 1994	0.1205
1,2-dibromoethane	B2	Tumors of nasal cavity	2.20E-04	IRIS, 1994	0.0045
acrylonitrile	B1	Respiratory cancers in humans	6.80E-05	IRIS, 1994	0.0147
cadmium(e)	B1	Lung/trachea/bronchus cancers	1.80E-03	IRIS, 1994	0.0006

(a) Median values as reported in RWDI (1992)

(b) For the chlorinated organics and benzene, concentrations were modeled by STB/OMOEE, July 95; for others, a dilution factor of 1.2E-4 was applied to the vent concentration; see text for details

(c) Based on data reported for chloromethane

(d) Based on oral RfD

(e) Mean concentration in airborne total particulates collected during 6 mnoths in the breathing zone of workers at 5 different landfills (Rahkonnen et al., 1987) - assumed to represent annual average concentration of cadmium.

Appendix B-III – B. Screening of Carcinogenic Air Emission Components (continued).

Results of the Screening Procedure for Carcinogens

Chemical	Concentration vent (a)	Maximum 24-h concentration at receptor (b)	Risk	Chemical score	Contribution
	ug/m3	ug/m3		%	%
Chlorinated Organics					
vinyl chloride	6467	0.774	6.50E-05	44.80	44.80
bromodichloromethane (e)	13803	1.656	3.31E-05	22.82	67.62
1,1-dichloroethylene	2975	0.36	1.80E-05	12.40	80.03
1,1,2,2-tetrachloroethane	1374	0.1656	9.60E-06	6.62	86.65
1,1-dichloroethane	38912	4.68	7.49E-06	5.16	91.81
methylene chloride	91780	10.98	5.16E-06	3.56	95.36
trichloroethylene	10716	1.278	2.56E-06	1.76	97.12
1,2-dichloroethane	445	0.054	1.40E-06	0.97	98.09
1,1,2-trichloroethane	544	0.0648	1.04E-06	0.71	98.81
chloroethane (d)	3110	0.378	6.80E-07	0.47	99.27
chloroform	190	0.0234	5.38E-07	0.37	99.65
chloromethane	1982	0.234	4.21E-07	0.29	99.94
carbon tetrachloride	50	0.00594	8.91E-08	0.06	100.00
			Total		
Others					
benzene	2180	0.26	2.16E-06	50.50	50.50
1,2-dibromoethane	54	0.00648	1.43E-06	33.36	83.86
acrylonitrile	84.3	0.010116	6.88E-07	16.10	99.96
cadmium(e)	0.0086	1.03E-06	1.86E-09	0.04	100.00

(a) Median values as reported in RWDI (1992)

(b) For the chlorinated organics and benzene, concentrations were modeled by STB/OMOEE, July 95; for others, a dilution factor of 1.2E-4 was applied to the vent concentration; see text for details

(c) Based on data reported for chloromethane

(d) Based on oral RfD

(e) Mean concentration in airborne total particulates collected during 6 months in the breathing zone of workers at 5 different landfills (Rahkonnen et al., 1987) - assumed to represent annual average concentration of cadmium.

Appendix B-III – C. Screening of Volatile Non-Carcinogenic Systemic Toxicants.

Screening of Non-carcinogenic Systemic Toxicants

Chemical	Critical end-point	RfC	Reference	Concentration (b)	Maximum 24-h Concentrations at Receptor Location (c)	Exposure Ratio	Total Exposure Ratio	Scores	Cumulative score
		ug/m3		ug/m3	ug/m3				
methyl mercaptan	IRR + CNS	1	IWA, 1994	20476	2.52	2.52000		54.6401	54.60
ethyl mercaptan	IRR	1	IWA, 1994	13693	1.638	1.63800		35.5160	90.12
hydrogen sulfide	inflammation nasal muc	1	IRIS, 1994	840	0.1008	0.10080	4.2588	2.1856	92.31
butan-2-ol (d)	hypoactivity & ataxia	300	IRIS, 1994	121000	14.4	0.04800		1.0408	93.35
nitrogen oxides	pulmonary function	100	HEAST, 1992	39700	4.764	0.04764		1.0330	94.38
dimethylsulphide (d)	NA	29	IWA, 1994	8710	0.97	0.03345		0.7252	95.10
hexane	CNS	200	IRIS, 1994	52700	6.324	0.03162		0.6856	95.79
pentane	SYS	115	NATICH, 1992	29600	3.552	0.03089		0.6697	96.46
tetrachloroethylene	multisite	100	CEPA, 1993	21600	2.592	0.02592		0.5620	97.02
1,2-dichloropropane	nasal muc hyperplasia	4	IRIS, 1994	789	0.09468	0.02367		0.5132	97.54
dichlorodifluoromethane (d)	reduced BW	600	IRIS, 1994	118000	14.16	0.02360		0.5117	98.05
toluene	CNS	400	IRIS, 1994	46900	5.628	0.01407		0.3051	98.35
octane	SYS	350	NATICH, 1992	36100	4.31	0.01231		0.2670	98.62
3-methylheptane	see octane	350	IWA, 1994	24900	2.988	0.00854		0.1851	98.80
butane	asphyxiant	620	NATICH, 1992	40600	4.872	0.00786		0.1704	98.97
THF	na (a)	80	MDEP, 1992	4740	0.5688	0.00711		0.1542	99.13
acetone (d)	increase liver, kidney W	300	IRIS, 1994	16400	1.968	0.00656		0.1422	99.27
dimethyldisulphide	squamous metaplasia	1.5	IWA, 1994	77	0.00924	0.00616		0.1336	99.40
decane	see nonane	1050	IWA, 1994	41500	4.98	0.00474		0.1028	99.51
MIK	liver & kidney effects	150	HEAST, 1992	5020	0.6024	0.00402		0.0871	99.59
dichlorofluoromethane	see diCldiFmethane	600	IRIS, 1994	18500	2.22	0.00370		0.0802	99.67
trichlorofluoromethane (d)	survival	900	IRIS, 1994	26000	3.12	0.00347		0.0752	99.75
1,1,1-trichloroethane (d)	liver	270	HEAST, 1992	6630	0.7956	0.00295		0.0639	99.81
lead	CNS	0.06	OMOEE, 1994	0.83	9.96E-05	0.00166		0.0360	99.85
1,4-dichlorobenzene	liver, kidney, proteinuria	240	CEPA, 1993	2720	0.3264	0.00136		0.0295	99.88
2,3-dimethylpentane	see octane	350	IWA, 1994	3290	0.3948	0.00113		0.0245	99.90
nonane	na	1050	NATICH, 1992	6320	0.7584	0.00072		0.0157	99.92
1,2-dichloroethylene (d)	increase in serum AKP	60	IRIS, 1994	354	0.04248	0.00071		0.0154	99.93
chromium	as Cr+3/NA	0.1	NYDOH, 1992	0.3	0.000036	0.00036		0.0078	99.94
butylacetate	IRR	700	IWA, 1994	1720	0.2064	0.00029		0.0064	99.95
iron	as Fe compounds/NA	1	NATICH, 1992	2.4	0.000288	0.00029		0.0062	99.95
chlorobenzene (d)	renal & hemato effects	25	CEPA, 1993	51	0.00612	0.00024		0.0053	99.96
chlorodifluoromethane	kid, adr, pitu	50000	IRIS, 1994	30200	3.624	0.00007		0.0016	99.96
copper	NA	2.4	CAPCOA, 1993	1.1	0.000132	0.00006		0.0012	99.96
styrene	CNS & BW	125	CEPA, 1993	28	0.00336	0.00003		0.0006	99.96
1,3-dichlorobenzene (d)	see o-dichlorobenzene	1310	NYDOH, 1992; IWA, 1994	115	0.0138	0.00001		0.0002	99.96
1,2-dichlorobenzene (d)	kidney	1310	CEPA, 1993	30	0.0036	0.00000		0.0001	99.96
zinc (d)	decreased SOD in RBC	900	IRIS, 1994	1.8	0.000216	0.00000	0.3532	0.0000	99.96
Total							4.612		

Appendix B-III – C. Screening of Volatile Non-Carcinogenic Systemic Toxicants.

Reproductive, Developmental and Related Toxicants (cont.)

*hydrogen sulfide	inflammation nasal muc	1	IRIS, 1994	840	0.1008	0.101	29.14	26.12
*methylene chloride	liver toxicity	183	IRIS, 1994	91780	10.98	0.060	17.35	43.47
*vinyl chloride	NA	26	CAPCOA, 1993	6467	0.774	0.030	8.61	52.08
*bromodichloromethane (d)	Kd cytomegaly	61	IRIS, 1994	13803	1.656	0.027	7.85	59.93
*tetrachloroethylene	multisite	100	CEPA, 1993	21600	2.592	0.026	7.49	67.42
*dichlorodifluoromethane (d)	dec BW	600	IRIS, 1994	118000	14.16	0.024	6.82	74.24
*toluene	CNS	400	IRIS, 1994	46900	5.628	0.014	4.07	78.31
*1,1-dichloroethane	NA	500	HEAST, 1992	38912	4.68	0.009	2.71	81.02
carbonyl sulphide	fetotox/terato	300	IRIS, 1994	20700	2.47	0.008	2.38	83.40
*acetone (d)	increase liver, kidney W	300	IRIS, 1994	16400	1.968	0.007	1.90	85.29
*1,1,2-trichloroethane	clin patho	12	IRIS, 1994	544	0.0648	0.005	1.56	86.85
cyclohexane	NA	280	NATICH, 1992	9680	1.1616	0.004	1.20	88.05
*benzene	NA	71	CAPCOA, 1993	2180	0.26	0.004	1.06	89.11
*trichlorofluoromethane (d)	survival	900	IRIS, 1994	26000	3.12	0.003	1.00	90.11
m-xylene	skeletal retardations	440	CEPA, 1993	11800	1.416	0.003	0.93	91.05
*carbon tetrachloride	liver lesions	2	IRIS, 1994	50	0.0059	0.003	0.86	91.90
*1,1,1-trichloroethane (d)	liver	270	HEAST, 1992	6630	0.7956	0.003	0.85	92.76
MEK	fetotox	1000	IRIS, 1994	18200	2.184	0.002	0.63	93.39
*trichloroethylene	NA	640	CAPCOA, 1993	10716	1.278	0.002	0.58	93.96
p-xylene	skeletal retardations	440	CEPA, 1993	5670	0.6804	0.002	0.45	94.41
o-xylene	skeletal retardations	440	CEPA, 1993	5280	0.6336	0.001	0.42	94.83
ethylbenzene	abs. of DEV effects	1000	IRIS, 1994	11800	1.416	0.001	0.41	95.24
*1,2-dibromoethane	NA	5	CA-OEHHA-PETS	54	0.0065	0.001	0.41	95.64
*1,4-dichlorobenzene	liver, kidney, proteinuria	240	CEPA, 1993	2720	0.3264	0.001	0.39	96.04
carbon disulphide	fetotox/terato	300	IRIS, 1994	2190	0.2628	0.001	0.25	96.29
*chloroform	liver toxicity	30	IRIS, 1994	190	0.0234	0.001	0.23	96.52
*1,2-dichloroethylene (d)	incr serum AKP	60	IRIS, 1994	354	0.0425	0.001	0.20	96.72
*1,2-dichloroethane	NA	95	CAPCOA, 1993	445	0.054	0.001	0.16	96.89
*chlorobenzene (d)	Kd & hemato effects	25	CEPA, 1993	51	0.0061	0.000	0.07	96.96
*chlorodifluoromethane	Kd, adren, pitu	50000	IRIS, 1994	30200	3.624	0.000	0.02	96.98
*1,3-dichlorobenzene (d)	see o-dichlorobenzene	1310	NYDOH, 1992	115	0.0138	0.000	0.00	96.98
*1,2-dichlorobenzene (d)	kidney	1310	CEPA, 1993	30	0.0036	0.000	0.00	96.98

(a) na: not availble

(b) Mean of values measured in the vent at the Brittania landfill as reported in RWDI, 1992 and IWA, 1994

(c) For the ethyl and methyl mercaptans, concentrations were modeled by STB/OMOEE, July 95; for other compounds, a dilution factor of 1.2E-4 was applied to the vent concentration; see

(d) RfC derived from an oral RfD by assuming 70 kg and 23 m3/day

*Exposure limit reported is for the most sensitive (critical) end-point

Appendix B-III – C. Screening of Volatile Non-Carcinogenic Systemic Toxicants (continued).

Screening of Chemicals Associated with Odour Issues

Chemical	Geometric Mean for Odour Detection Threshold (a)	Concentration (b)	Maximum 1-h Concentrations at Receptor Location	Maximum 10 minutes Concentrations at Receptor Location (e) Worst-Case	Exposure Ratio Worst-Case
	ug/m3	ug/m3	ug/m3	ug/m3	
ethyl mercaptan (d)	0.9	13693	12.1	72.6	80.67
methyl mercaptan	2.4	20476	17.6	105.6	44
octane	62	36100	30.9	185.4	2.99
butan-2-ol	591	121000	103.1	618.6	1.047
dimethylsulphide	51	8710	6.4454	38.6724	0.758
hydrogen sulfide	5.5	840	0.72	4.32	0.785
ethyl benzene	1900	11800	10.148 (c)	60.888	0.032
butylacetate	521	1720	1.4792 (c)	8.8752	0.017
MEK	4800	10716	9.21576 (c)	55.29456	0.012
dimethyldisulphide	66	77	0.06622 (c)	0.39732	0.006
MIK	6300	5020	4.3172 (c)	25.9032	0.004
carbon disulphide	3900	2190	1.8834 (c)	11.3004	0.003
acetone (d)	40000	16400	14.104 (c)	84.624	0.002
THF	18000	4740	4.0764 (c)	24.4584	0.001
decane	160000	41500	35.69 (c)	214.14	0.001
styrene	1300	28	0.02408 (c)	0.14448	1E-04
1,2-dichloropropane	40000	789	0.67854 (c)	4.07124	1E-04
chlorobenzene (d)	4500	51	0.04386 (c)	0.26316	6E-05

(a) Based on a OMOEE 1992 internal report prepared by Ortech International unless otherwise indicated

(b) Mean of values measured in the vent at the Brittania landfill as reported in RWDI, 1992 and IWA, 1994

(c) Estimated based on a dilution factor of 8.6E-04 obtained from atmospheric dispersion modelling

(d) 50% odour detection threshold based on ACGIH

(e) Obtained by multiplying the 1h-GLCmax by a factor of 6

Screening Approach for Heavy Metals Potentially Present in Particulates Emitted during the Operational Phase of the Landfill

List of Potential Heavy Metals Present in Particulates Emitted from Landfill (based on Rahkonien et al., 1987)

Air		End-points	
Cadmium	Cd	SYS	CARC
Chromium	Cr +3		SYS
Copper	Cu		SYS
Lead	Pb		SYS
Iron	Fe		SYS
Zinc	Zn		SYS

1) Estimated concentration in soil

Chemical	GLC at receptor location (a)		Dep	t1/2 in soil	Ks	Tf (b)	To (c)	Tt (d)	X	SD (e)	BD (f)	Cs			
	ug/m3	dep-rate	ug/m2-day	day	day-1	days	days	days		m	kg/m3	ug/kg (ppb)	year 22	year 24	year 30
Fe	1.90E-01	0.05	820.8	1.00E+08	6.93E-09	7300	0	7300	0.18	0.05	1333	44949.28	44949.05	44948.82	44948.14
Cd	4.30E-04		1.8576	1.00E+08	6.93E-09				0.18			101.73	101.73	101.73	101.72
Cr+3	1.50E-02		64.8	1.00E+08	6.93E-09				0.18			3548.63	3548.61	3548.59	3548.54
Pb	3.90E-02		168.48	1.00E+08	6.93E-09				0.18			9226.43	9226.38	9226.34	9226.20
Cu	7.30E-02		315.36	1.00E+08	6.93E-09				0.18			17269.99	17269.90	17269.81	17269.55
Zn	8.40E-02		362.88	1.00E+08	6.93E-09				0.18			19872.31	19872.21	19872.11	19871.81

(a) Estimated from literature data; for details see text
(b) End of accumulation period: 20 years
(c) Begining of evaluation period
(d) Total days of exposure
(e) Soil mixing depth; assumed to be 0.05 for both playground setting and gardening
(f) Soil bulk density

2) Estimated concentration in plants first 20 years

Chemical	GLC at receptor location (a)		Dep	IF (b)		Y (c)	D	K (d)	T (e)	Crop Dep Cdepv (f) Leafy	Cdepv Vines	Bio (g)	UF2 root crops (h)		UF2 leafy crops (h)	
	ug/m3	dep-rate	ug/m2-day			kg/m2	days	days-1	days	ug/kg	ug/kg					
Fe	1.90E-01	0.05	820.8	0.2	Leafy crops	2	14	0.0495	90	1638.91	819.46	1	4.00E-02	(j)	6.00E-02	(j)
Cd	4.30E-04		1.8576	0.1	Vines crops					3.71	1.85	1	4.00E-02		6.00E-02	
Cr+3	1.50E-02		64.8							129.39	64.69	1	1.00E-03	(j)	8.00E-04	(i)
Pb	3.90E-02		168.48							336.41	168.20	1	2.00E-03		5.00E-03	
Cu	7.30E-02		315.36							629.69	314.84	1	4.00E-02	(j)	6.00E-02	(j)
Zn	8.40E-02		362.88							724.57	362.29	1	4.00E-02	(j)	6.00E-02	(j)

(a) Estimated from literature data; for details see text
(b) Interception fraction by plants
(c) Crop yield
(d) Weathering constant
(e) Growth period
(f) Concentration due to direct deposition
(g) A bioavailability of 1 was assumed
(h) Default uptake factors as reported in CAPCOA (1993)
(i) Default value is for Cr+6 (CAOCOA, 1993)
(j) Values reported for Cd by CAPCOA (1993) were retained as surrrogate for this element based on their conservatism

Appendix B-III – D. Screening of Particulate-Bound Metals (continued).

2) Estimated concentration in plants first 20 years (cont.)

Chemical	UF2		Root uptake (j)	Root uptake (j)	Root uptake (j)	***Total concentration in crops***		
						Root Crops root uptake only	***Leafy Crops*** root uptake + deposition	***Vine Crops*** root uptake + deposition
	vine crops (h)		root crops	leafy crops	vine crops			
			ug/kg	ug/kg	ug/kg	ug/kg	ug/kg	ug/kg
Fe	2.00E-02	(j)	1797.97	2696.96	898.99	1797.97	4335.87	1718.44
Cd	2.00E-02		4.07	6.10	2.03	4.07	9.81	3.89
Cr+3	6.00E-04	(i)	3.55	2.84	2.13	3.55	132.23	66.82
Pb	1.00E-03		18.45	46.13	9.23	18.45	382.54	177.43
Cu	2.00E-02	(j)	690.80	1036.20	345.40	690.80	1665.89	660.24
Zn	2.00E-02	(j)	794.89	1192.34	397.45	794.89	1916.91	759.73

Appendix VI – D. Screening of Particulate-Bound Metals (continued).

3) Exposure by Inhalation

a) Inhalation variables associated with the receptors (extracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
IR (m3/d)	3.01	11.2	20.7	29.2	32
BW (kg)	6.28	13	26.8	50	61.5

b) Exposure by Inhalation to Estimated Concentrations of Heavy Metals over 20 Years

Chemicals	0-6m	7m-4y	5y-12y	13y-20y	21+	LADDpot	ADDpot (e)	Unit risk	RsC 10E-6	RfC
	ug/kg-day	ug/kg-day	ug/kg-day	ug/kg-day	ug/kg-day	ug/kg-day	ug/kg-day	(ug/m3)-1	ug/m3	ug/m3
Fe	9.11E-02	1.64E-01	1.47E-01	1.11E-01	9.89E-02		1.34E-01			1
Cd	2.06E-04	3.70E-04	3.32E-04	2.51E-04	2.24E-04	8.67E-05		1.80E-03	5.56E-04	
Cr+3	7.19E-03	1.29E-02	1.16E-02	8.76E-03	7.80E-03		1.06E-02			0.1
Pb(d)	1.87E-02	3.36E-02	3.01E-02	2.28E-02	2.03E-02		3.07E-02			NA
Cu	3.50E-02	6.29E-02	5.64E-02	4.26E-02	3.80E-02		5.15E-02			NA
Zn	4.03E-02	7.24E-02	6.49E-02	4.91E-02	4.37E-02		5.92E-02			2.4
I/O (c) 1										

(a) These values were calculated by considering receptor-specific inhalation rates and body weigths. In the case of Pb, the total Intake of Concern (IOC) was retained (OMOEE, 1994)

(b) Calculated by assuming 23 m3/day and 70 kg

(c) I/O based on Hawkins, 1991 based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)

(d) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y and compared to the IOC (1.85 ug/kg/day)

(e) ADDpot was estimated for the first 20 years of life (based on the operational lifetime of the landfill) and does not include the 20+ subgroup

Appendix B-III – D. Screening of Particulate-Bound Metals (continued).

3) Exposure by Inhalation (cont.)

b) Exposure by Inhalation to Estimated Concentrations of Heavy Metals over 20 Years (continued)

Chemicals	RdD ug/kg-day		Composite Receptor-based RsC 10E-6 or RfC (a) (ug/kg-day)	Conventional RsC 10E-6 and RfC (b) (ug/kg-day)	Ratio of Receptor-based to Conventional	ER	Risk on LADDpot
Fe		NATICH, 1992	5.73E-01	3.29E-01	1.74	0.2338	
Cd		IRIS, 1995	3.18E-04	1.83E-04	1.74	0.27212	2.7E-07
Cr+3		NYDOH, 1992	5.73E-02	3.29E-02	1.74	0.18458	
Pb(d)	1.85	OMOEE, 1994				0.01657	
Cu	300	IRIS, 1995				0.00017	
Zn		CAPCOA, 1993	1.38E+00			0.04307	

4) Exposure by Dermal Contact with Outdoor Soil

a) Dermal exposure variables associated with the receptors (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
SA (cm2) (assumed for active days) (b)	1747 (a)	2140	3370	4560	3840
BW (kg) (d)	6.28	13	26.8	50	61.5
SL (mg/cm2/day)	0.5				

(a) No data available in IWA (1994); value derived from data in Finley et al., (1994) by assuming a total skin surface of 4025 cm2 and maximum values for the fraction of arms, hands, and legs
(b) Based on IWA (1994)

b) Estimated exposure by dermal contact with soils (assumed to occur for 7 days/week x 8 months/year)

Chemicals	ABS	0-6m ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	13y-20y period 13-16 ug/kg-day	21+ period 21-25 ug/kg-day	ADDpot (d) ug/kg-day	RfD ug/kg-day (b)		ER
Fe	0.003 (a)	1.88E-02	1.11E-02	8.48E-03	6.15E-03	4.21E-03	4.58E-03	5.73E-01	NATICH, 1992	7.99E-03
Cd	0.003	4.24E-05	2.51E-05	1.92E-05	1.39E-05	9.53E-06	1.04E-05	1.00E+00	IRIS, 1995	1.04E-05
Cr+3	0.01 (a)	4.94E-03	2.92E-03	2.23E-03	1.62E-03	1.11E-03	1.20E-03	1.00E+00	IRIS, 1995	1.20E-03
Pb(c)	0.001	1.28E-03	7.59E-04	5.80E-04	4.21E-04	2.88E-04	4.34E-04	1.85E+00	OMOEE, 1994	2.35E-04
Cu	0.003 (a)	7.21E-03	4.26E-03	3.26E-03	2.36E-03	1.62E-03	1.76E-03	3.00E+02	IRIS, 1995	5.86E-06
Zn	0.003 (a)	8.29E-03	4.91E-03	3.75E-03	2.72E-03	1.86E-03	2.02E-03	1.01E+00	CAPCOA, 1993	2.00E-03

(a) The Dermal absorption factor for Cr+3 was assumed identical to Cr+6; values for Fe, Cu, and Zn were assumed identical to the dermal absorption factor for Cd

(b) The RfD for dermal exposure was assumed identical to the oral RfD, when available, under the assumption of diffusion to venous blood and hepatic processing prior to distribution to other organs and tissues. RfDs for Fe, Cr, and Zn were RfCs converted based on the composite receptor approach (see previous Table)

(c) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)

(d) ADDpot was estimated for 30 years according to the residence time at the site assuming similar concentrations of chemicals during the last 10 years
ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no contact with outdoor soil would occur during the first 3 months

Appendix B-III – D. Screening of Particulate-Bound Metals (continued).

5) Exposure by Ingestion of Soil of Outdoor Soil

a) Variables associated with the receptors for soil ingestion (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
Ing (mg soil/day) (a)	35	1300	1300	250	100
BW (kg)	6.28	13	26.8	50	61.5

(a) Based on IWA (1994)

b) Estimated exposure from soil ingestion (Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year)

Chemicals	ABS (a)	GI (b)	0-6m ug/kg-day	7m-4y ug/kg-day	5y-11y ug/kg-day	12y-19y ug/kg-day	20+ ug/kg-day	ADDpot (e) ug/kg-day	RfD (c) ug/kg-day		ER
Fe	1	1	2.51E-01	4.49E+00	2.18E+00	2.25E-01	7.31E-02	7.99E-01	5.73E-01	NATICH, 1992	1.39482
Cd	1	1	5.67E-04	1.02E-02	4.93E-03	5.09E-04	1.65E-04	1.81E-03	1.00E+00	IRIS, 1995	0.00181
Cr+3	1	1	1.98E-02	3.55E-01	1.72E-01	1.77E-02	5.77E-03	6.31E-02	1.00E+00	IRIS, 1995	0.06312
Pb(d)	1	1	5.14E-02	9.23E-01	4.48E-01	4.61E-02	1.50E-02	3.81E-01	1.85E+00	OMOEE, 1994	0.2061
Cu	1	1	9.62E-02	1.73E+00	8.38E-01	8.63E-02	2.81E-02	3.07E-01	3.00E+02	IRIS, 1995	0.00102
Zn	1	1	1.11E-01	1.99E+00	9.64E-01	9.94E-02	3.23E-02	3.53E-01	1.01E+00	CAPCOA, 1993	0.34995

(a) Fraction desorbed from soil particles and bioavailable for uptake

(b) Fraction absorbed through the GI tract

(c) RfDs for Fe, Cr, and Zn were RfCs converted based on the composite receptor approach (see Table on inhalation exposure)

(d) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)

(e) ADDpot was estimated for 30 years according to the residence time at the site assuming similar concentrations of chemicals during the last 10 years
ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no ingestion of outdoor soil would occur during the first 3 months

6) Exposure by Ingestion of Garden Produce

a) Variables associated with the receptors for ingestion of garden produce (extracted from IWA, 1994)

				0-6m		7m-4y		5y-12y		13y-20y		21+
Ingestion (Kg/day) (a)	Leafy crops			0.124		0.299		0.382		0.249		0.234
	Vine crops:	fruits	0.902	1.239	2.11	2.504	2.46	3.03	1.56	2.079	1.29	1.912
		vegetables	0.337		0.394		0.57		0.519		0.622	
	Root crops			0		0.49		0.795		0.404		0.414
BW (kg) (d)				6.28		13		26.8		50		61.5

(a) Based on IWA (1994)

b) Estimated exposure from ingestion of garden produce* (Exposure by ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year)

Chemicals	ABS (a)	GI (b)	UP(h)	0-6m (g) ug/kg-day	7m-4y (c) ug/kg-day	5y-11y ug/kg-day	12y-19y ug/kg-day	20+ ug/kg-day	ADDpot (f) ug/kg-day	RfD (d) ug/kg-day		ER
Fe	1	1	0.47	0.00E+00	1.24E+02	7.67E+01	2.68E+01	1.24E+01	3.10E+01	5.73E-01	NATICH, 1992	5.40E+01
Cd	1	1		0.00E+00	2.81E-01	1.73E-01	6.07E-02	2.80E-02	7.00E-02	1.00E+00	IRIS, 1995	7.00E-02
Cr+3	1	1		0.00E+00	4.05E+00	2.42E+00	8.75E-01	2.39E-02	9.10E-01	1.00E+00	IRIS, 1995	9.10E-01
Pb(e)	1	1		0.00E+00	1.11E+01	6.62E+00	2.39E+00	1.50E-01	2.51E+00	1.85E+00	OMOEE, 1994	1.36E+00
Cu	1	1		0.00E+00	4.77E+01	2.95E+01	1.03E+01	4.75E+00	1.19E+01	3.00E+02	IRIS, 1995	3.96E-02
Zn	1	1		0.00E+00	5.49E+01	3.39E+01	1.18E+01	5.47E+00	1.37E+01	1.01E+00	CAPCOA, 1993	1.35E+01

(a) Fraction desorbed from plants and bioavailable for uptake

(b) Fraction absorbed through the GI tract

(c) The fraction of plant type homegrown (i.e., of local origin) was estimated from data reported in USEPA (1990): leafy: 0.7, root crops: 0.2, vines: 0.5.

(d) RfDs for Fe, Cr, and Zn were RfCs converted based on the composite receptor approach (see Table on inhalation exposure)

(e) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)

(f) The contribution to ADDpot included deposition on leafy and vegetables plus root uptake for the first 20 years and only root uptake for the following 10 years

(g) The 0-6m subgroup was assumed to be breastfed and/or formula-fed

(h) Fraction of the total mass of ingested fruits and vegetables that consist of unprotected produce (McKone and Daniels, 1991)

Appendix B-III – D. Screening of Particulate-Bound Metals (continued).

Summary Table

	Inhalation - (L)ADDpot	Ingestion - ADD pot (ug/kg/day)		Dermal - ADDpot	Total Intake (ug/kg/day)	RfD	RsD	ER	Risk
Chemical	ug/kg/day	Soil	Garden produce	ug/kg/day		ug/kg/day	ug/kg/day		
Fe	1.34E-01	7.99E-01	3.10E+01	4.58E-03	3.19E+01	5.73E-01		55.63	
Cd	8.67E-05	-	-	-	8.67E-05		3.18E-04	0.27212	2.7E-07
		1.81E-03	7.00E-02	1.04E-05	7.19E-02	1.00E+00		0.07187	
Cr+3	1.06E-02	6.31E-02	9.10E-01	1.20E-03	9.85E-01	1.00E+00		0.98512	
Pb	3.07E-02	3.81E-01	2.51E+00	4.34E-04	2.92E+00	1.85E+00		1.578	
Cu	5.15E-02	3.07E-01	1.19E+01	1.76E-03	1.23E+01	3.00E+02		0.04084	
Zn	5.92E-02	3.53E-01	1.37E+01	2.02E-03	1.41E+01	1.01E+00		13.96	

1) Results of the Screening Procedure for Carcinogenic Compounds in the Leachate

Chemical	US EPA class	Critical end-point	Oral slope factor	Reference	Concentration in leachate (a)	Concentrations in groundwater (b)	Estimated intake (f)
			ug/(kg-day)E-1		ug/L	ug/L	ug/kg/day
Chlorinated Organics							
vinyl chloride	A	lung + liver tumors	5.26E-04	HEAST, 1992	120	1.71E-01	0.0037
1,1-dichloroethane	C	hemangiosarcomas	5.70E-06	CalEPA, 1996	670	4.80E+00	0.1029
1,1-dichloroethylene	C	adrenal pheocromocytomas	6.00E-04	IRIS, 1994	30	4.20E-02	0.0009
1,1,2,2-tetrachloroethane	C	hepatocellular carcinomas	2.00E-04	IRIS, 1994	17	1.80E-02	0.0004
1,2-dichloroethane	B2	hemangiosarcomas	9.10E-05	IRIS, 1994	20	2.80E-02	0.0006
bromodichloromethane	B2	kidney tumors	6.20E-05	IRIS, 1994	25	3.10E-02	0.0007
1,1,2-trichloroethane	C	hepatocellular carcinomas	5.70E-05	IRIS, 1994	25	3.00E-02	0.0006
chloromethane	C	kidney tumors	1.30E-05	HEAST, 1992	50	7.00E-02	0.0015
methylene chloride	B2	adenomas + carcinomas	7.50E-06	IRIS, 1994	4000	5.00E-02	0.0011
chloroform	B2	kidney tumors	6.10E-06	IRIS, 1994	20	2.70E-02	0.0006
chloroethane (c)	C	kidney tumors	1.30E-05	IWA, 1994	100	5.90E-03	0.0001
trichloroethylene (e)	NA	NA	6.00E-06	CAPCOA, 1993	80	4.30E-04	9E-06
bromochloromethane (d)	B2	adenomas + carcinomas	7.50E-06	IRIS, 1994	0.2	2.80E-04	6E-06
Others							
benzene		Leukemia	2.90E-05	IRIS, 1994	25	2.60E-02	0.0006
pentachlorophenol		hemangiosarcomas	1.20E-04	IRIS, 1994	1	8.10E-07	2E-08
2,4,6-trichlorophenol		leukemia	1.10E-05	IRIS, 1994	5	2.20E-06	5E-08

(a) Maximum annual geometric mean concentration from representative landfills under reasonable worst case conditions
(b) Maximum predicted concentrations in the groundwater aquifer at the receptor location under reasonable worst case conditions
(c) Based on data reported for chloromethane
(d) Based on data reported for methylene chloride
(e) Oral slope factor was based on the unit risk factor reported in CAPCOA (1993) by assuming 23 cu. m/day and 70 kg
(f) Calculated by assuming an ingestion rate of 1.5 L/day (tapwater + tapwater-based beverages) and a body weight of 70 Kg (CEPA, 1993)

1) Results of the Screening Procedure for Carcinogenic Compounds in the Leachate (continued)

Chemical	Estimated risk	Total risk	Scores	Cumulative scores
			%	%
Chlorinated Organics				
vinyl chloride	2E-06	2E-06	58.5	58.5
1,1-dichloroethane	6E-07		17.79	76.29
1,1-dichloroethylene	5E-07		16.38	92.67
1,1,2,2-tetrachloroethane	8E-08		2.34	95.01
1,2-dichloroethane	5E-08		1.66	96.67
bromodichloromethane	4E-08		1.25	97.92
1,1,2-trichloroethane	4E-08		1.11	99.03
chloromethane	2E-08		0.59	99.62
methylene chloride	8E-09		0.24	99.86
chloroform	4E-09		0.11	99.97
chloroethane (d)	2E-09		0.05	100.02
trichloroethylene (f)	6E-11		1.7E-02	100.04
bromochloromethane (e)	5E-11	1.3E-06	1.4E-02	100.05
		3.3E-06		
Others				
benzene	2E-08		99.984	98.96
pentachlorophenol	2E-12		0.0129	98.98
2,4,6-trichlorophenol	5E-13	1.6E-08	0.0032	98.98

2) Results of the Screening Procedure for Systemic Toxicants in the Leachate

Chemical	Critical end-point/ organ	RfD	Reference	Concentration in leachate (c)	Concentration in groundwater (d)	Estimated intake (e)
		ug/kg/day		ug/L	ug/L	ug/kg/day
Nitrate	methemoglobinemia	1600	IRIS, 1994	7.00E+04	2.00E+03	42.8571
1,2-dichloroethylene	incr in serum AKP	60	IRIS, 1994	3.00E+01	7.50E-01	0.01607
isovaleric acid	NA	2500	IWA, 1994	1.82E+05	1.10E+01	0.23571
phenyl acetic acid	NA	143	IWA, 1994	1.06E+04	6.20E-01	0.01329
Butyric acid (a)	NA	2500	IWA, 1994	1.50E+05	9.30E+00	0.19929
Hexanoic acid	NA	4000	IWA, 1994	3.00E+05	1.40E+01	0.3
isobutyric acid (a)	NA	2500	IWA, 1994	8.60E+04	5.40E+00	0.11571
dimethylsulphide	NA	29	IWA, 1994	2.20E+03	2.90E-02	0.00062
o & p-cresol	NA	60	CAPCOA, 1993	1.20E+03	5.30E-02	0.00114
propionic acid	NA	40000	IWA, 1994	2.60E+05	1.60E+01	0.34286
1,1,1-trichloroethane	liver	270	HEAST, 1992	5.00E+01	6.50E-02	0.00139
toluene	CNS	400	IRIS, 1994	1.23E+03	4.00E-02	0.00086
acetic acid	NA	14300	IWA, 1994	8.60E+04	1.20E+00	0.02571
1,4-dichlorobenzene	liver, kidney	240	CEPA, 1993	2.00E+01	6.80E-03	0.00015
Octanoic acid	Diarrhea	400	IWA, 1994	1.00E+04	5.00E-03	0.00011
heptanoic acid (b)	NA	4000	IWA, 1994	2.20E+04	4.00E-02	0.00086
ethyl benzene	kidney, liver	100	IRIS, 1994	2.90E+02	5.50E-04	1.2E-05
tetrachloroethylene	multisite	100	CEPA, 1993	5.50E+01	4.00E-04	8.6E-06

(a) Assumed as isovaleric acid
(b) Assumed as hexanoic acid
(c) Maximum annual geometric mean concentration from representative landfills under reasonable worst case conditions
(d) Maximum predicted concentrations in the groundwater aquifer at the receptor location under reasonable worst case conditions
(e) Calculated by assuming an ingestion rate of 1.5 L/day (tapwater + tapwater-based beverages) and a body weight of 70 Kg (CEPA, 1993)

2) Results of the Screening Procedure for Systemic Toxicants in the Leachate (continued)

Chemical	Exposure ratio	Scores	Cumulative scores
		%	%
Nitrate	0.0268	97.399	97.40
1,2-dichloroethylene	0.0003	0.974	98.37
isovaleric acid	9E-05	0.3428	98.72
phenyl acetic acid	9E-05	0.3378	99.05
Butyric acid (a)	8E-05	0.2899	99.34
Hexanoic acid	8E-05	0.2727	99.62
isobutyric acid (a)	5E-05	0.1683	99.78
dimethylsulphide	2E-05	0.0779	99.86
o & p-cresol	2E-05	0.0688	99.93
propionic acid	9E-06	0.0312	99.96
1,1,1-trichloroethane	5E-06	0.0188	99.98
toluene	2E-06	0.0078	99.99
acetic acid	2E-06	0.0065	100.00
1,4-dichlorobenzene	6E-07	0.0022	100.00
Octanoic acid	3E-07	0.001	100.00
heptanoic acid (b)	2E-07	0.0008	100.00
ethyl benzene	1E-07	0.0004	100.00
tetrachloroethylene	9E-08	0.0003	100.00
Total	0.0275		

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

Sample Calculation for the Estimation of Concentrations in Soil

a) Concentration in soil at end of year 4 for emissions 1-4/Playground setting

Chemical	in the vent (a) Concentrations	Maximum annual average Concentration at receptor location 0-4 (b)		Dep (c)	t1/2 in soil (d)	Ks
	ug/m3	ug/m3	dep-rate	ug/m2-day	day	day-1
1,1,2,2-tetrachloroethane	1374	0.0054	6.33E-03	2.94	44	1.58E-02
bromodichloromethane	13803	0.0537		29.36	180	3.85E-03
1,1-dichloroethane	38912	0.1503		82.20	154	4.50E-03
methylene chloride	91780	0.3527		192.90	50	1.39E-02
1,1-dichloroethylene	2975	0.115		62.91	180	3.85E-03
vinyl chloride	6467	0.0253		13.84	180	3.85E-03
trichloroethylene	10716	0.0414		22.65	43	1.61E-02
1,2-dichloroethane	445	0.0017		0.92	90	7.70E-03
1,1,2-trichloroethane	544	0.0021		1.17	365	1.90E-03
chloroethane	3110	0.0121		6.63	50	1.39E-02
benzene	2180	0.0084		4.60	16	4.33E-02
hydrogen sulfide	840	0.0032		NA	NA	NA
ethyl mercaptan	13693	0.0537		NA	NA	NA
methyl mercaptan	20476	0.0798		NA	NA	NA
Total		0.9				

(a) Median values as reported in RWDI (1992)
(b) Based on concentrations at end of year 4; values were assumed to be lognormally distributed with a standard deviation of 3.5
(c) Based on a deposition rate of 600 m/day on plant surfaces (McKone and Daniels, 1991)
(d) Based on ranges of values (min and max) reported in MacKay et al. (1992); assumed to be uniformely distributed
(e) Not available; value is the t1/2 for dibromochloromethane
(f) End of accumulation period: 4 years

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

a) Concentration in soil at end of year 4 for emissions 1-4/Playground setting (continued)

Total concentration at year 4

Chemical	Tf (f)	To (g)	Tt (h)	X	SD (i)	BD (j)		T	Tend	Cs
	days	days	days		m	kg/m3	Distribution	days	days	ug/kg (ppb)
1,1,2,2-tetrachloroethane	1460	0	1460	1396.51	0.05	1800	uniform	1460	0	1.98
bromodichloromethane				1201.20						69.71
1,1-dichloroethane				1238.09						172.12
methylene chloride				1387.85						147.00
1,1-dichloroethylene				1201.20						149.38
vinyl chloride				1201.20						32.86
trichloroethylene				1397.95						14.95
1,2-dichloroethane				1330.13						1.21
1,1,2-trichloroethane				966.24						4.55
chloroethane				1387.85						5.05
benzene				1436.91						1.16
hydrogen sulfide				NA						NA
ethyl mercaptan				NA						NA
methyl mercaptan				NA						NA

(g) Begining of evaluation period
(h) Total days of exposure
(i) Soil mixing depth; assumed to be 0.05 for both playground setting and gardening
(j) Soil bulk density; uniform distribution with 1000-1800 kg/m3; based on Copeland et al.,1994

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

Summary Table - Total concentration of carcinogens in soil (Cs) at different periods - in ug/kg or ppb

Chemical	End of year 4	End of year 8 Year 6*	End of year 12 Year 10*	End of year 16 Year 14*	End of year 20 Year 18*	End of year 25 Year 22.5*	End of year 30 Year 27.5*	End of year 40 Year 35*
	ug/kg	ug/kg	ug/kg	ug/kg	ug/kg	ug/kg	ug/kg	ug/kg
1,1,2,2-tetrachloroethane	1.98	2.94	3.22	4.38	6.58	2.94	2.77	1.90
bromodichloromethane	69.71	107.47	119.38	160.85	241.01	113.84	103.80	73.69
1,1-dichloroethane	172.12	261.45	12.88	370.55	591.33	280.07	242.65	178.35
methylene chloride	147.00	218.77	248.16	323.56	493.06	221.68	211.44	138.10
1,1-dichloroethylene	149.38	31.02	26.82	33.74	52.33	24.79	22.45	15.62
vinyl chloride	32.86	50.17	57.59	53.13	47.66	35.37		
trichloroethylene	14.95	22.97	25.16	32.70	49.71	22.37	21.21	14.14
1,2-dichloroethane	1.21	1.83	2.09	2.72	4.19	1.93	1.77	1.22
1,1,2-trichloroethane	4.55	7.76	9.19	11.81	17.37	10.01	8.32	5.62
chloroethane	5.05	7.51	8.47	10.79	16.98	7.39	7.02	4.83
benzene	1.16	1.71	1.95	2.53	3.87	1.73	1.62	1.09

*Midway of the time interval 5-8, 9-12, 13-16, 17-20, 21-25, 26-30, and 31-40 used for the calculation of residual chemicals in soil

Sample Calculations for the Estimation of Concentrations in Plants (deposition + root uptake)

a) Concentration in plants for emissions 1-4 year

Chemical	Maximum annual average concentration at receptor location 0-4 (a)		Dep (b)	IF (c)		Y (d)	Vda	R	T	Kow (h)	H (h)	Rv (e)
	ug/m3	dep-rate	ug/m2-day			kg/m2	m/day	torr-L/mol-K	K		torr-L/mol	days-1
1,1,2,2-tetrachloroethane	0.0054	6.33E-03	2.94	0.39	Leafy crops	4.4	547	62.4	298	245	347	91
bromodichloromethane	0.0537		29.4	0.073	Vines crops					126	1679	831
1,1-dichloroethane	0.1503		82.2							62	4462	4201
methylene chloride	0.3527		193							18	1860	4605
1,1-dichloroethylene	0.115		62.9							135	144367	67015
vinyl chloride	0.0253		13.8							14	42592	123784
trichloroethylene	0.041		22.7							263	8888	2184
1,2-dichloroethane	0.0017		0.92							30	998	1711
1,1,2-trichloroethane	0.0021		1.17							118	691	364
chloroethane	0.0121		6.63							27	8587	15944
benzene	0.0084		4.6							135	4178	1939

(a) Based on concentrations at end of year 4; for detail see text
(b) Based on a deposition rate of 600 m/day on plant surfaces (McKone and Daniels, 1991)
(c) Interception fraction by plants (Copeland et al., 1994)
(d) Crop yield (Copeland et al., 1994)
(e) Weathering constant Ke replaced by a quantitative component that integrates evaporation from produce surfaces (McKone and Daniels, 1991); for details see text
(f) Growth period (Copeland et al., 1994)
(g) Concentration due to direct deposition
(h) Henry's Law constant (H) - Organic Carbon partition coefficient (Koc); value shown is the mode of an assumed triangular distribution based on values reported in MacKay et al. (1992) unless otherwise indicated (* uniform distribution based on 2 values)
(i) Uptake factor based on soil concentration
(j) Concentration due to root translocation or uptake-to be used to calculate intake of chemicals for potatoes
(k) Average concentration in and on leafy and vine crops

a) Concentration in plants for emissions 1-4 year

		Crop Dep				Root uptake (for Deposition Only Potatoes)		
Chemical	T (f)	Cdepv (g) Leafy	Cdepv Vines	Koc (h)	UF2 (i)	Ctrans (j)	Cf leafy (k)	Cf Vines (k)
	days	ug/kg	ug/kg			ug/kg	ug/kg	ug/kg
1,1,2,2-tetrachloroethane	90	5.32E-01	9.95E-02	79	0.37	0	0.00285	0.00053
bromodichloromethane		5.83E-02	1.09E-02 *	251	0.08	0	0.00313	0.00059
1,1-dichloroethane		1.15E-02	2.16E-03	40	0.38	0	0.00173	0.00033
methylene chloride		1.05E-02	1.97E-03	17	0.65	0	0.00371	0.0007
1,1-dichloroethylene		7.23E-04	1.35E-04 *	150	0.14	0	8.32E-05	1.56E-05
vinyl chloride		3.92E-04	7.33E-05	32	0.33	0	9.91E-06	1.86E-06
trichloroethylene		2.22E-02	4.15E-03	158	0.19	0	0.00092	0.00017
1,2-dichloroethane		2.83E-02	5.30E-03	19	0.65	0	4.78E-05	8.95E-06
1,1,2-trichloroethane		1.33E-01	2.49E-02	70	0.29	0	0.00029	5.36E-05
chloroethane		3.04E-03	5.69E-04	29	0.41	0	3.68E-05	6.90E-06
benzene		2.50E-02	4.68E-03	83	0.26	0	0.00021	3.93E-05

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

1) Exposure through Inhalation

a) Inhalation variables associated with the receptors (extracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
IR (m3/d)	1.21	11	15.9	26.1	21.4
BW (kg)	6.28	13	26.8	50	61.5

IR was correlated to BW, r = 0.67

b) Exposure through Inhalation of Carcinogens

Chemicals	0-6m ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	5y-12y period 9-12 ug/kg-day	13y-20y period 13-16 ug/kg-day	13y-20y period 17-20 ug/kg-day	21+ period 21-25 ug/kg-day	21+ period 26-30 ug/kg-day	LADDpot ug/kg-day
1,1,2,2-tetrachloroethane	8.27E-04	3.63E-03	3.77E-03	4.14E-03	4.95E-03	7.45E-03	2.20E-03	2.07E-03	1.65E-03
bromodichloromethane	8.27E-03	3.63E-02	3.77E-02	4.14E-02	4.95E-02	7.45E-02	2.20E-02	2.07E-02	1.65E-02
1,1-dichloroethane	2.32E-02	1.02E-01	1.06E-01	1.38E-03	1.35E-01	2.11E-01	6.35E-02	5.59E-02	3.97E-02
methylene chloride	5.44E-02	2.39E-01	2.49E-01	2.83E-01	3.24E-01	4.94E-01	1.47E-01	1.40E-01	1.10E-01
1,1-dichloroethylene	1.77E-02	7.79E-02	8.05E-03	9.31E-03	1.04E-02	1.62E-02	4.79E-03	4.47E-03	7.19E-03
vinyl chloride	3.90E-03	1.71E-02	1.76E-02	2.00E-02	2.30E-02	3.48E-02	1.03E-02	9.49E-03	7.75E-03
trichloroethylene	6.38E-03	2.80E-02	3.02E-02	3.31E-02	3.78E-02	5.75E-02	1.71E-02	1.62E-02	1.29E-02
1,2-dichloroethane	2.60E-04	1.14E-03	1.21E-03	1.38E-03	1.58E-03	2.43E-03	7.33E-04	6.71E-04	5.36E-04
1,1,2-trichloroethane	3.31E-04	1.45E-03	1.48E-03	1.69E-03	1.94E-03	2.91E-03	8.79E-04	8.38E-04	6.56E-04
chloroethane	1.87E-03	8.20E-03	8.56E-03	9.65E-03	1.08E-02	1.70E-02	4.88E-03	4.64E-03	3.73E-03
	I/O***= 0.75								
benzene	1.30E-03	5.69E-03	5.88E-03	6.69E-03	7.64E-03	1.17E-02	3.47E-03	3.26E-03	2.60E-03

* This RsC was calculated as for the LADDpot by considering receptor-specific inhalation rates and body weigths

** Calculated by assuming 23 m3/day and 70 kg

*** the indoor/outdoor, I/O, ratio for long-term exposure was assumed to fit a triangular distribution ranging from 0.2 to 1(Hawkins, 1991) with a most likeliest value of 0.75 as determined for particulates (Hawley, 1985)

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

b) Exposure through Inhalation of Carcinogens

Unit risk (ug/m3)-1	RsC 10E-6 ug/m3	Composite Receptor-based RsC 10E-6* (ug/kg-day)	Conventional RsC 10E-6** (ug/kg-day)	Ratio of Receptor-based to Conventional	ER	Risk	
5.80E-05	1.72E-02	7.23E-03	5.67E-03	1.28E+00	2.29E-01	2.29E-07	1,1,2,2-tetrachloroethane
2.00E-05	5.00E-02	2.10E-02	1.64E-02	1.28E+00	7.88E-01	7.88E-07	bromodichloromethane
1.60E-06	6.25E-01	2.51E-01	2.05E-01	1.28E+00	6.25E-02	6.53E-08	1,1-dichloroethane
4.70E-07	2.13E+00	8.93E-01	6.99E-01	1.28E+00	1.23E-01	1.23E-07	methylene chloride
5.00E-05	2.00E-02	8.39E-03	6.57E-03	1.28E+00	8.57E-01	8.57E-07	1,1-dichloroethylene
8.40E-05	1.19E-02	4.99E-03	3.91E-03	1.28E+00	1.55E+00	1.55E-06	vinyl chloride
2.00E-06	5.00E-01	2.10E-01	1.64E-01	1.28E+00	6.15E-02	6.15E-08	trichloroethylene
2.60E-05	3.85E-02	1.61E-02	1.26E-02	1.28E+00	3.32E-02	3.32E-08	1,2-dichloroethane
1.60E-05	6.25E-02	2.62E-02	2.05E-02	1.28E+00	2.50E-02	2.50E-08	1,1,2-trichloroethane
1.80E-06	5.56E-01	2.33E-01	1.83E-01	1.28E+00	1.60E-02	1.60E-08	chloroethane
				total	3.75E+00	3.75E-06	
8.30E-06	1.20E-01	5.05E-02	3.96E-02	1.28E+00	5.14E-02	5.14E-08	benzene

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

2) Exposure through Dermal Contact with Soil - Carcinogens

a) Dermal exposure variables associated with the receptors (extracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
SA (cm2) (assumed for active days) (b)	1747(a)	2140	2380	3630	3400
BW (kg) (d)	6.28	13	26.8	50	61.5
SL (mg/cm2/day) (c)	0.52				

SA Variables for the 0-6m

%		Total SA/kg	
0.2	641	1747	Arms
0.1			Hands
0.2			Legs

(a) No data available in IWA (1994); value derived from data in Finley et al., (1994) by using the triangular distributions for arms, hands, and legs, the distribution for surface area/body weight ratio for 0-2 infants and BW distribution from IWA (1994)

(b) Based on IWA (1994); SA was for active days

(c) Based on Finley et al. (1994)

(d) A correlation of 0.67 was assumed between BW and SA

b) Calculated exposure through dermal contact with soils * (Exposure through dermal contact with soils was assumed to occur for 7 days/week x 8 months/year)

Chemicals	Kow	Kh (a)	ABS (b)	0-6m (d) ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	5y-12y period 9-12 ug/kg-day	13y-20y period 13-16 ug/kg-day	13y-20y period 17-20 ug/kg-day	21+ period 21-25 ug/kg-day
1,1-dichloroethane	66	0.298	0.03	7.47E-04	4.42E-04	3.62E-04	1.78E-05	4.20E-04	6.70E-04	2.42E-04
vinyl chloride	14	3.21	0.03	1.43E-04	8.44E-05	6.95E-05	7.98E-05	8.46E-05	1.28E-04	4.58E-05
1,1-dichloroethylene	99	2.35	0.03	6.48E-04	3.84E-04	4.30E-05	3.72E-05	3.82E-05	5.93E-05	2.14E-05
Bromodichloromethane	106	0.309	0.03	3.02E-04	1.79E-04	1.49E-04	1.65E-04	1.82E-04	2.73E-04	9.82E-05
1,1,2,2-tetrachloroethane	386	0.024	0.03	8.60E-06	5.09E-06	4.07E-06	4.45E-06	4.96E-06	7.46E-06	2.54E-06
methylene chloride	20	0.128	0.03	6.38E-04	3.77E-04	3.03E-04	3.44E-04	3.66E-04	5.58E-04	1.91E-04
Trichloroethylene	531	0.304	0.03	6.49E-05	3.84E-05	3.18E-05	3.49E-05	3.70E-05	5.63E-05	1.93E-05
1,2-dichloroethane	34	0.035	0.03	5.26E-06	3.11E-06	2.54E-06	2.90E-06	3.08E-06	4.75E-06	1.67E-06
1,1,2-trichloroethane	146	0.028	0.03	1.97E-05	1.17E-05	1.08E-05	1.27E-05	1.34E-05	1.97E-05	8.64E-06
Chloroethane	27	0.729	0.03	2.19E-05	1.30E-05	1.04E-05	1.17E-05	1.22E-05	1.92E-05	6.37E-06
Benzene	152	0.222	0.03	5.04E-06	2.98E-06	2.37E-06	2.70E-06	2.86E-06	4.39E-06	1.49E-06

(a) Calculated based on Kh=P/(SxRxT) (see McKone, 1990)

(b) Fraction absorbed accross the skin; uptake in viable tissues reflective of a potential dose (see McKone, 1990); a maximal value of 0.03 was assumed based on the recommendation of McKone (1990)

(c) The slope factor for dermal exposure to carcinogens was assumed identical to the oral slope factor, when available, under the assumption of diffusion to venous blood and hepatic processing prior to distributio to other organs and tissues

(d) ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no contact with outdoor soil would occur during the first 3 months

b) Calculated exposure through dermal contact with soils * (Exposure through dermal contact with soils was assumed to occur for 7 days/week x 8 months/year)

21+ Period 26-30 Ug/kg-day	LADDpot ug/kg-day	Oral slope factor (c) (ug/kg-day)E-1	RsD 10E-6 (ug/kg-day)	ER	Risk	
2.09E-04	9.53E-05	5.70E-06	1.75E-01	5.12E-04	5.12E-10	1,1-dichloroethane
4.11E-05	2.13E-05	5.26E-04	1.90E-03	1.12E-02	1.12E-08	vinyl chloride
1.94E-05	2.39E-05	6.00E-04	1.67E-03	1.43E-02	1.43E-08	1,1-dichloroethylene
8.95E-05	4.55E-05	6.20E-05	1.61E-02	2.82E-03	2.82E-09	bromodichloromethane
2.39E-06	1.24E-06	2.00E-04	5.00E-03	2.48E-04	2.48E-10	1,1,2,2-tetrachloroethane
1.82E-04	9.30E-05	7.50E-06	1.33E-01	6.97E-04	6.97E-10	methylene chloride
1.83E-05	9.44E-06	6.00E-06	1.67E-01	5.67E-05	5.67E-11	trichloroethylene
1.53E-06	7.84E-07	9.10E-05	1.10E-02	7.14E-05	7.14E-11	1,2-dichloroethane
7.17E-06	3.38E-06	5.70E-05	1.75E-02	1.93E-04	1.93E-10	1,1,2-trichloroethane
6.05E-06	3.16E-06	1.30E-05	7.69E-02	4.11E-05	4.11E-11	chloroethane
			Total	2.66E-02 Total	2.66E-08	
1.40E-06	7.28E-07	2.90E-05	3.45E-02	2.11E-05	2.11E-11	benzene

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

3) Exposure through Ingestion of Soil - Carcinogens

a) Variables associated with the receptors for soil ingestion (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
Ing (mg soil/day) (a)	35	91	91	250	100
BW (kg) (b)	6.28	13	26.8	50	61.5

(a) Based on IWA (1994); Ing and BW were correlated at r= 0.67
(b) Based on IWA (1994)

b) Calculated exposure from soil ingestion * * Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year

Chemicals	ABS (a)	GI (b)	0-6m (c) ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	5y-12y period 9-12 ug/kg-day	13y-20y period 13-16 ug/kg-day	13y-20y period 17-20 ug/kg-day	21+ period 21-25 ug/kg-day	21+ period 26-30 ug/kg-day
1,1-dichloroethane	1	1	9.59E-04	1.20E-03	8.88E-04	4.37E-05	1.85E-03	2.96E-03	4.55E-04	3.95E-04
vinyl chloride	1	1	1.83E-04	2.30E-04	1.70E-04	1.96E-04	3.74E-04	5.63E-04	8.64E-05	7.75E-05
1,1-dichloroethylene	1	1	8.33E-04	1.05E-03	1.05E-04	9.11E-05	1.69E-04	2.62E-04	4.03E-05	3.65E-05
bromodichloromethane	1	1	3.89E-04	4.88E-04	3.65E-04	4.05E-04	8.04E-04	1.21E-03	1.85E-04	1.69E-04
1,1,2,2-tetrachloroethane	1	1	1.10E-05	1.39E-05	9.97E-06	1.09E-05	2.19E-05	3.29E-05	4.78E-06	4.50E-06
methylene chloride	1	1	8.19E-04	1.03E-03	7.43E-04	8.43E-04	1.62E-03	2.47E-03	3.60E-04	3.44E-04
trichloroethylene	1	1	8.33E-05	1.05E-04	7.80E-05	8.54E-05	1.63E-04	2.49E-04	3.64E-05	3.45E-05
1,2-dichloroethane	1	1	6.76E-06	8.49E-06	6.23E-06	7.11E-06	1.36E-05	2.10E-05	3.14E-06	2.88E-06
1,1,2-trichloroethane	1	1	2.53E-05	3.18E-05	2.64E-05	3.12E-05	5.91E-05	8.68E-05	1.63E-05	1.35E-05
chloroethane	1	1	2.81E-05	3.53E-05	2.55E-05	2.88E-05	5.39E-05	8.49E-05	1.20E-05	1.14E-05
benzene	1	1	6.47E-06	8.13E-06	5.81E-06	6.61E-06	1.26E-05	1.94E-05	2.81E-06	2.64E-06

(a) Fraction desorbed from soil particles and bioavailable for uptake
(b) Fraction absorbed through the GI tract
(c) ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no ingestion of outdoor soil would occur during the first 3 months

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

b) Calculated exposure from soil ingestion * * Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year

LADDpot ug/kg/day	Oral slope factor (c) (ug/kd-day)E-1	RsD 10E-6 (ug/kg-day)	ER	Risk	
3.04E-04	5.70E-06	1.75E-01	1.7E-03	1.72E-09	1,1-dichloroethane
6.61E-05	5.26E-04	1.90E-03	3.48E-02	3.48E-08	vinyl chloride
6.57E-05	6.00E-04	1.67E-03	3.94E-02	3.94E-08	1,1-dichloroethylene
1.41E-04	6.20E-05	1.61E-02	8.75E-03	8.75E-09	bromodichloromethane
3.85E-06	2.00E-04	5.00E-03	7.69E-04	7.69E-10	1,1,2,2-tetrachloroethane
2.88E-04	7.50E-06	1.33E-01	2.16E-03	2.16E-09	methylene chloride
2.92E-05	6.00E-06	1.67E-01	1.75E-04	1.75E-10	trichloroethylene
2.43E-06	9.10E-05	1.10E-02	2.21E-04	2.21E-10	1,2-dichloroethane
1.04E-05	5.70E-05	1.75E-02	5.91E-04	5.91E-10	1,1,2-trichloroethane
9.80E-06	1.30E-05	7.69E-02	1.27E-04	1.27E-10	chloroethane
		Total	8.9E-02	8.64E-08	
2.26E-06	2.90E-05	3.45E-02	6.54E-05	6.54E-11	benzene

4) Exposure through Ingestion of Garden Produce - Carcinogens

a) Variables associated with the receptors for ingestion of garden produce (extracted from IWA, 1994)

			0-6m	7m-4y	5y-12y	13y-20y	21+
Ingestion (Kg/day) (a)	Leafy crops		0.015	0.0372	0.0475	0	0
	Vine crops:	fruits	0.112 0.154	0.3 0.311	0.305 0.376	0.287 0.4	0.237 0.4
		vegetables	0.042	0	0.071	0.095	0.114
	Root crops		0	0.061	0.099	0.1	0.1
BW (kg) (d)			6.28	13	26.8	50	61.5

(a) Based on IWA (1994); Ing for each produce category was correlated to BW (r = 0.67)

b) Calculated exposure from ingestion of garden produce* * Exposure through ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year

Chemicals	ABS (a)	GI (b)	UP (d)	0-6m (e) ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	5y-12y period 9-12 ug/kg-day	13y-20y period 13-16 ug/kg-day	13y-20y period 17-20 ug/kg-day	21+ period 21-25 ug/kg-day	21+ period 26-30 ug/kg-day
1,1-dichloroethane	1	1	0.47	0	3.5E-06	3.1E-06	4E-08	2.4E-06	3.7E-06	1.3E-06	1.1E-06
vinyl chloride	1	1		0	2E-08	1.7E-08	2E-08	1.4E-08	2.1E-08	6.9E-09	6.4E-09
1,1-dichloroethylene	1	1		0	1.7E-07	1.5E-08	1.7E-08	1.1E-08	1.8E-08	6E-09	5.6E-09
Bromodichloromethane	1	1		0	6.2E-06	8.6E-06	6.1E-06	4.4E-06	6.6E-06	2.2E-06	2.1E-06
1,1,2,2-tetrachloroethane	1	1		0	5.7E-06	5.1E-06	5.6E-06	4E-06	6E-06	2E-06	1.9E-06
methylene chloride	1	1		0	7.4E-06	6.6E-06	7.5E-06	5.2E-06	7.9E-06	2.7E-06	2.5E-06
Trichloroethylene	1	1		0	1.8E-06	1.7E-06	1.9E-06	1.3E-06	1.9E-06	6.5E-07	6.2E-07
1,2-dichloroethane	1	1		0	9.5E-08	8.7E-08	9.9E-08	6.8E-08	1E-07	3.6E-08	3.3E-08
1,1,2-trichloroethane	1	1		0	5.7E-07	5E-07	5.7E-07	3.9E-07	5.9E-07	2E-07	1.9E-07
Chloroethane	1	1		0	7.3E-08	6.6E-08	7.4E-08	5E-08	7.9E-08	2.6E-08	2.4E-08
Benzene	1	1		0	4.2E-07	3.7E-07	4.2E-07	2.9E-07	4.5E-07	1.5E-07	1.4E-07

(a) Fraction desorbed from plants and bioavailable for uptake
(b) Fraction absorbed through the GI tract
(c) The fraction of plant type homegrown (i.e., of local origin) was estimated from data reported in USEPA (1990): leafy: 0.7, root crops: 0.2, vines: 0.5.
(d) Fraction of the total mass of ingested fruits and vegetables that consist of unprotected produce (McKone and Daniels, 1991)
(e) The 0-6m subgroup was assumed to be breast-fed and/or formula-fed

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

b) Calculated exposure from ingestion of garden produce* * Exposure through ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year

LADDpot ug/kg-day	Oral slope factor (c) (ug/kg-day)E-1	RsD 10E-6 (ug/kg-day)	ER	Risk	
5.8E-07	0.0000057	0.175	7.3E-06	7.34E-12	1,1-dichloroethane
4E-09	0.000526	0.001901	2.1E-06	2.13E-12	vinyl chloride
8.4E-09	0.0006	0.001667	5.1E-06	5.07E-12	1,1-dichloroethylene
1.4E-06	0.000062	0.016129	8.7E-05	8.68E-11	bromodichloromethane
1.2E-06	0.0002	0.005	0.00023	2.34E-10	1,1,2,2-tetrachloroethane
1.5E-06	7.5E-06	0.133333	1.2E-05	1.16E-11	methylene chloride
3.8E-07	0.000006	0.166667	2.3E-06	2.29E-12	trichloroethylene
2E-08	0.000091	0.010989	1.8E-06	1.84E-12	1,2-dichloroethane
1.2E-07	0.000057	0.017544	6.6E-06	6.65E-12	1,1,2-trichloroethane
1.5E-08	0.000013	0.076923	2E-07	1.97E-13	chloroethane
		Total	0.0003 Total	3.58E-10	
8.7E-08	0.000029	0.034483	2.5E-06	2.51E-12	benzene

5) Exposure through Ingestion of Groundwater

a) Ingestion variables associated with the receptors (extracted from IWA, 1994)

	0-6 m	7 m - 4 y	5 y - 12 y	13 y - 20 y	21 + y
IR (L/day)	0.8	0.83	1	1	1.51

Ir was correlated to BW, r = 0.67

b) Exposure through Ingestion of Groundwater containing vinyl chloride

	Maximum PredictedConcentration in groundwater (ug/L)	0-6 m	7 m - 4 y	5 y - 12 y	13 y - 20 y	21 + y	LADDpot	Risk slope factor (ug/kg-day)E-1	Risk
Reasonable worst case	0.171	2.18E-02	1.09E-02	6.38E-03	3.42E-03	4.20E-03	2.39E-03	5.26E-04	1.26E-06
Anticipated conditions	5.5E-05	7.01E-06	3.51E-06	2.05E-06	1.10E-06	1.35E-06	7.69E-07	5.26E-04	4.04E-10

Appendix B-III – F. Results of the Detailed Assessment for Carcinogenic Volatiles (continued).

Summary Table for All Pathways
Risks Associated with Exposure to carcinogenic chemicals at the maximum GLC - Deterministic Approach

Chemical	Inhalation vapours		Ingestion Soil		Ingestion Garden produce		Dermal	Soil		Total Risk per chemical (a)
Chlorinated Compounds										
		%		%		%			%	
1,1,2,2-tetrachloroethane	2.3E-07	99.46	7.69E-10	0.33	2.339E-10	0.1		2.48E-10	0.11	2.3E-07
Bromodichloromethane	7.9E-07	98.54	8.75E-09	1.09	8.677E-11	0.01		2.82E-09	0.35	8E-07
1,1-dichloroethane	6.5E-07	98.51	1.72E-09	1.13	7.34E-12	0.00		5.12E-10	0.35	6.72E-08
methylene chloride	1.2E-07	97.72	2.16E-09	1.71	1.156E-11	0.01		6.97E-10	0.55	1.26E-07
1,1-dichloroethylene	8.6E-07	94.10	3.94E-08	4.33	5.068E-12	0.00		1.43E-08	1.58	9.11E-07
vinyl chloride	1.6E-06	97.12	3.48E-08	2.18	2.128E-12	0.00		1.12E-08	0.7	1.6E-06
Trichloroethylene	6.1E-08	99.62	1.75E-10	0.28	2.29E-12	0.00		5.67E-11	0.09	6.17E-08
1,2-dichloroethane	3.3E-08	99.12	2.21E-10	0.66	1.838E-12	0.01		7.14E-11	0.21	3.35E-08
1,1,2-trichloroethane	2.5E-08	96.94	5.91E-10	2.29	6.65E-12	0.03		1.93E-10	0.75	2.58E-08
chloroethane	1.6E-08	98.96	1.27E-10	0.79	1.968E-13	0.00		4.11E-11	0.25	1.62E-08
Sub-total			8.64E-08		3.58E-10					
										Total for all routes
Total inhalation risk	3.75E-06	96.98%	Total ingestion risk		8.68E-08	2.25%	Total dermal	3.02E-08	0.76%	3.95E-06
Others										Total for all routes
Benzene			6.54E-11		2.5E-12					
Total inhalation risk	5.14E-08	99.83%	Total ingestion risk		6.79E-11	0.13%	Total dermal	2.11E-11	0.04%	5.15E-08

(a) The conceptual assumption of additivitiy across different routes may not be valid for this compound due to differences in route-specific target organs and end-points

Appendix B-III – G. Results of the Detailed Assessment for Volatiles Exhibiting Systemic Effects.

Determination of Exposure Ratios for Systemic Toxicants

	Vent concentrations	GLC at the Point of Maximum Concentration (ug/m3)							Rfd	Maximum ER (a)
Chemical	ug/m3	0-4	5-8	9-12	13-16	17-20	21-25	26-30	ug/m3	Year 17-20
hydrogen sulfide	840	0.003	0.0048	0.005423	0.007037	0.0108	0.0048	0.0045	1	0.01
ethyl mercaptan	13693	0.054	0.0795	0.087132	0.118603	0.176694	0.079	0.07431	1	0.18
methyl mercaptan	20476	0.08	0.1166	0.130698	0.172514	0.271529	0.1176	0.11046	1	0.27

Exposure Ratios Associated with Odourous Compounds

Chemical	Geometric Mean for Odour Detection Threshold (a)	Maximum 1-h Concentrations at Receptor Location (c) Year 1-4	Maximum 10-min Concentrations at Receptor Location (c) Year 1-4	Exposure Ratio	Maximum 1-h Concentrations at Receptor Location (c) Year 5-8	Maximum 10-min Concentrations at Receptor Location (c) Year 5-8	Exposure Ratio
	ug/m3	ug/m3			ug/m3		
ethyl mercaptan (d)	0.9	3.02	9.1	10	2	4.5	5
methyl mercaptan	2.4	2.03	6.1	3	2	6.6	2.75
octane	62	3.87	12	0	5	15.78	0.2545161
butane-2-ol	591	12.9	39	0	18	52.8	0.0893401
dimethylsulphide	51	1.19	3.6	0	1	2.622	0.0514118
hydrogen sulfide	5.5	0.12	0.4	0	0	0.27	0.0490909

(a) Based on a OMOEE 1992 internal report prepared by Ortech International unless otherwise indicated

(b) Mean of values measured in the vent at the Brittania landfill as reported in RWDI, 1992 and IWA, 1994

(c) Modelled concentration at maximum ground levels concentration

(d) 50% odour detection threshold based on ACGIH

Appendix B-III – G. Results of the Detailed Assessment for Volatiles Exhibiting Systemic Effects.

Exposure Ratios Associated with Odourous Compounds (cont.)

Chemical	Maximum 1-h Concentration at Receptor location (c) Year 9-12	Maximum 10-min Concentration at Receptor location (c) Year 9-12	Exposure ratio	Maximum 1-h Concentration at Receptor location (c) Year 13-16	Maximum 10-min Concentration at Receptor location (c) Year 13-16	Exposure ratio	Maximum 1-h Concentration at Receptor location (c) Year 17-20	Maximum 10-min Concentration at Receptor location (c) Year 17-20	Exposure ratio
	ug/m3			ug/m3			ug/m3		
ethyl mercaptan	6.24	19	20.8	12.1	36	40	5	16.2	18
methyl mercaptan	9.36	28	11.7	17.6	53	22	8	24.78	10.325
octane	16.7	50	0.808	30.9	93	1	14	42.3	0.682
butane-2-ol	56	168	0.284	103.1	309	1	47	141.3	0.239
dimethylsulphide	3.8	11	0.224	6.98	21	0	3	9.57	0.187
hydrogen sulfide	0.39	1.2	0.213	0.72	2.2	0	0	0.99	0.18

Exposure Ratios Associated with Odourous Compounds (cont.)

Chemical	Maximum 1-h Concentrations at Receptor Location (c) Year 21-25	Maximum 10-min Concentrations at Receptor Location (c) Year 21-25	Exposure Ratio	Maximum 1-h Concentrations at Receptor Location (c) Year 26-30	Maximum 10-min Concentrations at Receptor Location (c) Year 26-30	Exposure Ratio
	ug/m3			ug/m3		
ethyl mercaptan	8.1	24.3	27	2.26	7	7.533
methyl mercaptan	12	36.3	15	3.36	10	4.200
octane	21	63.6	1	5.87	18	0.284
butane-2-ol	71	211.8	0.4	19.6	59	0.099
dimethylsulphide	4.8	14.34	0.3	1.33	4	0.078
hydrogen sulfide	0.5	1.47	0.3	0.14	0	0.076

Probabilistic Approach for Heavy Metals Potentially Present in Particulates Emitted during the Operational Phase of the Landfill

List of Potential Heavy Metals Present in Particulates Emitted from Landfill (based on Rahkonien et al., 1987)

Air			End-points
Cadmium	Cd	SYS	CARC
Chromium	Cr +3		SYS
Copper	Cu		SYS
Lead	Pb		SYS
Iron	Fe		SYS
Zinc	Zn		SYS

1) Estimated concentration in soil during the 20 years of emission

Chemical	GLC at receptor location (a)		Dep (b)	t1/2 in soil	Ks	Tf (c)	To (d)	Tt (e)	X	SD (f)
	ug/m3	dep-rate	ug/m2-day	day	day-1	days	days	days		m
Fe	1.90E-01	6.30E-03	103.4208	1.00E+08	6.93E-09	7300	0	7300	0.18	0.05
Cd	4.30E-04		0.2340576	1.00E+08	6.93E-09				0.18	
Cr+3	1.50E-02		8.1648	1.00E+08	6.93E-09				0.18	
Pb	3.90E-02		21.22848	1.00E+08	6.93E-09				0.18	
Cu	7.30E-02		39.73536	1.00E+08	6.93E-09				0.18	
Zn	8.40E-02		45.72288	1.00E+08	6.93E-09				0.18	

(a) Estimated from literature data; for details see text
(b) Based on a deposition rate of 0.0063 m/sec (McKone and Daniels, 1990)
(c) End of accumulation period: 20 years
(d) Begining of evaluation period
(e) Total days of exposure
(f) Soil mixing depth; assumed to be 0.05 for both playground setting and gardening
(g) Soil bulk density; uniform distribution with 1000-1800 kg/m3; based on Copeland et al., (1994)

Appendix B-III – H. Results of the Detailed Assessment for Particle-bound Metals.

1a) Estimated concentration in soil during the 20 years of emission

Chemical	BD (g) kg/m3	Cs ug/kg (ppb)
Fe	1800	4194.22
Cd		9.49
Cr+3		331.12
Pb		860.92
Cu		1611.46
Zn		1854.29

1b) Estimated concentration in soil during the remaining 10 years of decay

year 22	year 24	year 26	year 28	year 30
4194.20	4194.17			4194.11
9.49	9.49			9.49
331.12	331.12			331.11
860.91	860.91			860.90
1611.45	1611.45			1611.42
1854.28	1854.27			1854.24

Appendix B-III – H. Results of the Detailed Assessment for Particle-bound Metals.

2) Estimated concentration in plants

										Crop Dep		
Chemical	GLC at receptor location (a)		Dep (b)	IF (c)		Y (d)	D	K (e)	T (f)	Cdepv (g) Leafy	Cdepv Vines	Bio (h)
	ug/m3	dep-rate	ug/m2-day			kg/m2	days	days-1	days	ug/kg	ug/kg	
Fe	1.90E-01	6.30E-03	103.42	0.39	Leafy crops	4.4	10	0.0693	90	132.02	24.71	1
Cd	4.30E-04		0.23	0.073	Vines crops					0.30	0.06	1
Cr+3	1.50E-02		8.16							10.42	1.95	1
Pb	3.90E-02		21.23							27.10	5.07	1
Cu	7.30E-02		39.74							50.72	9.49	1
Zn	8.40E-02		45.72							58.37	10.92	1

(a) Estimated from literature data; for details see text
(b) Based on a deposition rate of 0.0063 m/sec (McKone and Daniels, 1990)
(c) Interception fraction by plants (Copeland et al., 1994)
(d) Crop yield (Copeland et al., 1994)
(e) Weathering constant
(f) Growth period (Copeland et al., 1994)
(g) Concentration due to direct deposition
(h) A bioavailability of 1 was assumed
(i) Default uptake factors as reported in CAPCOA (1993)
(j) Default value is for Cr+6 (CAOCOA, 1993)
(k) Values reported for Cd by CAPCOA (1993) were retained as surrrogate for this element based on their conservatism

2) Estimated concentration in plants

Chemical	UF2 root crops (i)	UF2 leafy crops (i)	UF2 vine crops (i)	**Root uptake (j)** root crops	**Root uptake (j)** leafy crops	**Root uptake (j)** vine crops
				ug/kg	ug/kg	ug/kg
Fe	4.00E-02 (k)	6.00E-02 (k)	2.00E-02 (k)	167.77	251.65	83.88
Cd	4.00E-02	6.00E-02	2.00E-02	0.38	0.57	0.19
Cr+3	1.00E-03 (j)	8.00E-04 (j)	6.00E-04 (j)	0.33	0.26	0.20
Pb	2.00E-03	5.00E-03	1.00E-03	1.72	4.30	0.86
Cu	4.00E-02 (k)	6.00E-02 (k)	2.00E-02 (k)	64.46	96.69	32.23
Zn	4.00E-02 (k)	6.00E-02 (k)	2.00E-02 (k)	74.17	111.26	37.09

Total concentration in crops

Chemical	***Root Crops*** root uptake only	***Leafy Crops*** root uptake + deposition	***Vine Crops*** root uptake + deposition
	ug/kg	ug/kg	ug/kg
Fe	167.77	383.67	108.60
Cd	0.38	0.87	0.25
Cr+3	0.33	10.69	2.15
Pb	1.72	31.40	5.93
Cu	64.46	147.41	41.72
Zn	74.17	169.62	48.01

3) Exposure by Inhalation

a) Inhalation variables associated with the receptors (extracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
IR (m3/d)	1.21	11	15.9	26.1	21.4
BW (kg)	6.28	13	26.8	50	61.5

A correlation of 0.67 was assumed between inhalation rate and body weight

b) Exposure by Inhalation to Estimated Concentrations of Heavy Metals over 20 Years

Chemicals	0-6m ug/kg-day	7m-4y ug/kg-day	5y-12y ug/kg-day	13y-20y ug/kg-day	21+ ug/kg-day	LADDpot ug/kg-day
Fe	2.93E-02	1.29E-01	9.02E-02	7.93E-02	5.29E-02	
Cd	6.63E-05	2.91E-04	2.04E-04	1.80E-04	1.20E-04	7.60E-05
Cr+3	2.31E-03	1.02E-02	7.12E-03	6.26E-03	4.18E-03	
Pb(d)	6.01E-03	2.64E-02	1.85E-02	1.63E-02	1.09E-02	
Cu	1.13E-02	4.94E-02	3.46E-02	3.05E-02	2.03E-02	
Zn	1.29E-02	5.69E-02	3.99E-02	3.51E-02	2.34E-02	
I/O (c)	0.75					

(a) These values were calculated using receptor-specific inhalation rates and body weigths. In the case of Pb, the total Intake of Concern (IOC) was retained (OMOEE, 1994)

(b) Calculated by assuming 23 m3/day and 70 kg

(c) I/O based on Hawkins, 1991 and Hawlwy, 1985

(d) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y and compared to the IOC (1.85 ug/kg/day) based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)

(e) ADDpot was estimated for the first 20 years of life (based on the operational lifetime of the landfill) and does not include the 20+ subgroup

b) Exposure by Inhalation to Estimated Concentrations of Heavy Metals over 20 Year

Chemicals	ADDpot (e) ug/kg-day	Unit risk (ug/m3)-1	RsC 10E-6 ug/m3	RfC ug/m3	RfD ug/kg-day		Composite Receptor-based RsC 10E-6 or RfC (a) (ug/kg-day)
Fe	9.10E-02			1		NATICH, 1992	4.20E-01
Cd		1.80E-03	5.56E-04			IRIS, 1995	2.33E-04
Cr+3	7.19E-03			0.1		NYDOH, 1992	4.20E-02
Pb(d)	2.03E-02			NA	1.85	OMOEE, 1994	
Cu	3.50E-02			NA	300	IRIS, 1995	
Zn	4.03E-02			2.4		CAPCOA, 1993	1.01E+00

Chemicals	Conventional RsC 10E-6 and RfC (b) (ug/kg-day)	Ratio of Receptor-based to Conventional	ER	Risk on LADDpot
Fe	3.29E-01	1.28	0.2169	
Cd	1.83E-04	1.28	0.3258	3E-07
Cr+3	3.29E-02	1.28	0.1713	
Pb(d)			0.011	
Cu			0.0001	
Zn			0.04	

4) Exposure by Dermal Contact with Soil

a) Dermal exposure variables associated with the receptors (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
SA (cm2) (assumed for active days) (b)	1746.66 (a)	2140	2380	3630	3400
BW (kg) (d)	6.28	13	26.8	50	61.5
SL (mg/cm2/day) (c)	0.52				

(a) No data available in IWA (1994); value derived from data in Finley et al., (1994) by using the triangular distributions for arms, hands, and legs, the distribution for surface area/body weight ratio for 0-2 infants and BW distribution from IWA (1994)

(b) Based on IWA (1994)
(c) Based on Finley et al. (1994)
(d) A correlation of 0.67 was assumed between BW and SA

b) Estimated exposure by dermal contact with soils (assumed to occur for 7 days/week x 8 months/year)

Chemicals	ABS	0-6m ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	13y-20y period 13-16 ug/kg-day	21+ period 21-25 ug/kg-day	ADDpot ug/kg-day
Fe	0.003 (a)	1.82E-03	1.08E-03	5.81E-04	4.75E-04	3.62E-04	3.69E-04
Cd	0.003	4.12E-06	2.44E-06	1.32E-06	1.08E-06	8.19E-07	8.35E-07
Cr+3	0.01 (a)	4.79E-04	2.83E-04	1.53E-04	1.25E-04	9.52E-05	9.71E-05
Pb(c)	0.001	1.25E-04	7.37E-05	3.98E-05	3.25E-05	2.47E-05	3.48E-05
Cu	0.003 (a)	6.99E-04	4.14E-04	2.23E-04	1.83E-04	1.39E-04	1.42E-04
Zn	0.003 (a)	8.05E-04	4.76E-04	2.57E-04	2.10E-04	1.60E-04	1.63E-04

(a) The Dermal absorption factor for Cr+3 was assumed identical to Cr+6; values for Fe, Cu, and Zn were assumed identical to the dermal absorption factor for Cd

(b) The RfD for dermal exposure was assumed identical to the oral RfD, when available, under the assumption of diffusion to venous blood and hepatic processing prior to distribution to other organs and tissues. RfDs for Fe, Cr, and Zn were RfCs converted based on the composite receptor approach (see previous Table)

(c) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)

(d) ADDpot was estimated for 30 years according to the residence time at the site assuming similar concentrations of chemicals during the last 10 years ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no contact with outdoor soil would occur during the first 3 months

Appendix B-III – H. Results of the Detailed Assessment for Particle-bound Metals.

b) Estimated exposure by dermal contact with soils (assumed to occur for 7 days/week x 8 months/year)

SA Variables for the 0-6m

	%	Total SA/kg	SA (cm2)
Arms	0.151	641	1746.66
Hands	0.0539		
Legs	0.229		

Chemicals	RfD ug/kg-day (b)		ER
Fe	4.20E-01	NATICH, 1992	8.79E-04
Cd	1.00E+00	IRIS, 1995	8.35E-07
Cr+3	1.00E+00	IRIS, 1995	9.71E-05
Pb(c)	1.85E+00	OMOEE, 1994	1.88E-05
Cu	3.00E+02	IRIS, 1995	4.72E-07
Zn	1.01E+00	CAPCOA, 1993	1.61E-04

5) Exposure by Ingestion of Soil

a) Variables associated with the receptors for soil ingestion (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
Ing (mg soil/day) (a)	35	91	91	250	100
BW (kg)	6.28	13	26.8	50	61.5

(a) Based on IWA (1994) A correlation of 0.67 was assumed between BW and SA

b) Estimated exposure from soil ingestion (Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year)

Chemicals	ABS (a)	GI (b)	0-6m ug/kg-day	7m-4y ug/kg-day	5y-11y ug/kg-day	12y-19y ug/kg-day	20+ ug/kg-day
Fe	1	1	2.34E-02	2.94E-02	1.42E-02	2.10E-02	6.82E-03
Cd	1	1	5.29E-05	6.64E-05	3.22E-05	4.75E-05	1.54E-05
Cr+3	1	1	1.85E-03	2.32E-03	1.12E-03	1.66E-03	5.38E-04
Pb(d)	1	1	4.80E-03	6.03E-03	2.92E-03	4.30E-03	1.40E-03
Cu	1	1	8.98E-03	1.13E-02	5.47E-03	8.06E-03	2.62E-03
Zn	1	1	1.03E-02	1.30E-02	6.30E-03	9.27E-03	3.02E-03

(a) Fraction desorbed from soil particles and bioavailable for uptake
(b) Fraction absorbed through the GI tract
(c) RfDs for Fe, Cr, and Zn were RfCs converted based on the composite receptor approach (see Table on inhalation exposure)
(d) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y
based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)
(e) ADDpot was estimated for 30 years according to the residence time at the site assuming similar concentrations of chemicals during the last 10 years. ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no ingestion of outdoor soil would occur during the first 3 months

Appendix B-III – H. Results of the Detailed Assessment for Particle-bound Metals.

b) Estimated exposure from soil ingestion **(Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year)**

Chemicals	ADDpot ug/kg-day	RfD (c) ug/kg-day		ER
Fe	1.03E-02	4.20E-01	NATICH, 1992	0.0246
Cd	2.33E-05	1.00E+00	IRIS, 1995	2E-05
Cr+3	8.13E-04	1.00E+00	IRIS, 1995	0.0008
Pb(d)	2.58E-03	1.85E+00	OMOEE, 1994	0.0014
Cu	3.96E-03	3.00E+02	IRIS, 1995	1E-05
Zn	4.56E-03	1.01E+00	CAPCOA, 1993	0.0045

6) Exposure by Ingestion of Garden Produce

a) Variables associated with the receptors for ingestion of garden produce (extracted from IWA, 1994)

			0-6m		7m-4y		5y-12y		13y-20y		21+
Ingestion (Kg/day) (a)											
Leafy crops			0.0154		0.0372		0.0475		0.0456		0.0428
Vine crops:	fruits	0.112	0.154	0.262	0.311	0.305	0.376	0.287	0.382	0.237	0.351
	vegetables	0.042		0.049		0.071		0.095		0.114	
Root crops			0		0.061		0.099		0.074		0.076
BW (kg) (d)			6.28		13		26.8		50		61.5

(a) Based on IWA (1994) A correlation of 0.67 was assumed between BW and ingestion rates

b) Estimated exposure from ingestion of garden produce* (Exposure by ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year)

Chemicals	ABS (a)	GI (b)	UP(h)	0-6m (c) ug/kg-day	7m-4y ug/kg-day	5y-11y ug/kg-day	12y-19y ug/kg-day	20+ ug/kg-day
Fe	1	1	0.47	0.00E+00	1.13E+00	7.06E-01	3.60E-01	2.12E-01
Cd	1	1		0.00E+00	2.56E-03	1.60E-03	8.14E-04	4.79E-04
Cr+3	1	1		0.00E+00	2.25E-02	1.36E-02	7.16E-03	4.09E-04
Pb(e)	1	1		0.00E+00	6.45E-02	3.91E-02	2.06E-02	2.57E-03
Cu	1	1		0.00E+00	4.34E-01	2.71E-01	1.38E-01	8.13E-02
Zn	1	1		0.00E+00	4.99E-01	3.12E-01	1.59E-01	9.35E-02

(a) Fraction desorbed from plants and bioavailable for uptake
(b) Fraction absorbed through the GI tract
(c) The fraction of plant type homegrown (i.e., of local origin) was estimated from data reported in USEPA (1990): leafy: 0.7, root crops: 0.2, vines: 0.5.
(d) RfDs for Fe, Cr, and Zn were RfCs converted based on the composite receptor approach (see Table on inhalation exposure)
(e) A ADDpot for lead was computed for a composite group including 0-6m, 7m-4y and 5y-11y based on evidence that these represent the most sensitive subgroups to the neurotoxic effects induced by lead (OMOEE, 1994)
(f) The contribution to ADDpot included deposition on leafy and vegetables plus root uptake for the first 20 years and only root uptake for the following 10 years
(g) The 0-6m subgroup was assumed to be breastfed and/or formula-fed
(h) Fraction of the total mass of ingested fruits and vegetables that consist of unprotected produce (McKone and Daniels, 1991)

b) Estimated exposure from ingestion of garden produce* (Exposure by ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year)

Chemicals	ADDpot ug/kg-day	RfD (d) ug/kg-day		ER
Fe	3.26E-01	4.20E-01	NATICH, 1992	7.76E-01
Cd	7.38E-04	1.00E+00	IRIS, 1995	7.38E-04
Cr+3	5.55E-03	1.00E+00	IRIS, 1995	5.55E-03
Pb(e)	3.01E-02	1.85E+00	OMOEE, 1994	1.63E-02
Cu	1.25E-01	3.00E+02	IRIS, 1995	4.17E-04
Zn	1.44E-01	1.01E+00	CAPCOA, 1993	1.43E-01

Summary Table

Chemical	Inhalation - (L)ADDpot ug/kg/day	Soil (ug/kg/day)	Ingestion - ADD pot Garden produce (ug/kg/day)	Dermal - ADDpot ug/kg/day	Total Intake (ug/kg/day)	RfD ug/kg/day	RsD ug/kg/day	ER	Risk
Fe	9.10E-02	1.03E-02	3.26E-01	3.69E-04	4.28E-01	4.20E-01		1.0188	
Cd	7.60E-05	-	-	-	7.60E-05		2.33E-04	0.3258	3.26E-07
		2.33E-05	7.38E-04	8.35E-07	7.62E-04	1.00E+00		0.0008	
Cr+3	7.19E-03	8.13E-04	5.55E-03	9.71E-05	1.36E-02	1.00E+00		0.0136	
Pb	2.03E-02	2.58E-03	3.01E-02	3.48E-05	5.30E-02	1.85E+00		0.0287	
Cu	3.50E-02	3.96E-03	1.25E-01	1.42E-04	1.64E-01	3.00E+02		0.0005	
Zn	4.03E-02	4.56E-03	1.44E-01	1.63E-04	1.89E-01	1.01E+00		0.1872	

1) Concentration in soil / Playground setting

chemical	Concentrations at the vent (a)	as ratio of highest concentration	Maximum annual average concentration at receptor location 0-4 (b)		Dep (c)	t1/2 in soil (d)	Ks	Tf (f)	To (g)	Tt (h)	X
	ug/m3		ug/m3	dep-rate	ug/m2-day	day	day-1	days	days		
1,1,2,2-tetrachloroethane	1374	0.0353	1	6.33E-03	546.91	44	0.0158	10950	0	10950	10887
bromodichloromethane	13803	0.3547	1		546.91	180	0.0039				10690
1,1-dichloroethane	38912	1	1		546.91	154	0.0045				10728
methylene chloride	91780	2.3587	1		546.91	50	0.0139				10878
1,1-dichloroethylene	2975	0.0765	1		546.91	180	0.0039				10690
vinyl chloride	6467	0.1662	1		546.91	180	0.0039				10690
trichloroethylene	10716	0.2754	1		546.91	43	0.0161				10888
1,2-dichloroethane	445	0.0114	1		546.91	90	0.0077				10820
1,1,2-trichloroethane	544	0.014	1		546.91	365	0.0019				10423
chloroethane	3110	0.0799	1		546.91	50	0.0139				10878
benzene	2180	0.056	1		546.91	16	0.0433				10927
hydrogen sulfide	840	0.0216	1		NA	NA	NA				NA
ethyl mercaptan	13693	0.3519	1		NA	NA	NA				NA
methyl mercaptan	20476	0.5262	1		NA	NA	NA				NA

(a) Median values as reported in RWDI (1992)
(b) Based on concentrations at end of year 4; values were assumed to be lognormally distributed with a standard deviation of 3.5
(c) Based on a deposition rate of 600 m/day on plant surfaces (McKone and Daniels, 1991)
(d) Based on ranges of values (min and max) reported in MacKay et al. (1992); assumed to be uniformely distributed
(e) Not available; value is the t1/2 for dibromochloromethane
(f) End of accumulation period: 4 years
(g) Begining of evaluation period
(h) Total days of exposure
(i) Soil mixing depth; assumed to be 0.05 for both playground setting and gardening
(j) Soil bulk density; uniform distribution with 1000-1800 kg/m3; based on Copeland et al.,1994

1) Concentration in soil / Playground setting (cont.)

	SD (i)	BD (j)		T	Tend	Cs
	m	kg/m3	Distribution	days	days	ug/kg (ppb)
1,1,2,2-tetrachloroethane	0.05	1800	uniform			383.59
bromodichloromethane						1540.9
1,1-dichloroethane						1323
methylene chloride						435.55
1,1-dichloroethylene						1540.9
vinyl chloride						1540.9
trichloroethylene						374.92
1,2-dichloroethane						779.83
1,1,2-trichloroethane						3046.7
chloroethane						435.55
benzene						140.01
hydrogen sulfide						NA
ethyl mercaptan						NA
methyl mercaptan						NA

2) Concentration in plants

Chemical	Maximum annual average concentration at receptor location 0-4 (a)		Dep (b)	IF (c)		Y (d)	Vda	R	T	Kow (h)	H (h)	Rv (e)	T (f)
	ug/m3	dep-rate	ug/m2-day			kg/m2	m/day	torr-L/mol-K	K		torr-L/mol	days-1	days
1,1,2,2-tetrachloroethane	0.0353	6.33E-03	546.91	0.39	Leafy crops	4.4	546.91	62.4	298	245	346.5	91.187	90
bromodichloromethane	0.3547		546.91	0.073	Vines					126	1679	831.34	
1,1-dichloroethane	1		546.91							62	4462	4200.8	
methylene chloride	2.3587		546.91							18	1860	4604.8	
1,1-dichloroethylene	0.0765		546.91							135	144367	67015	
vinyl chloride	0.1662		546.91							14	42592	123784	
trichloroethylene	0.2754		546.91							263	8888	2184.2	
1,2-dichloroethane	0.0114		546.91							30	998	1710.5	
1,1,2-trichloroethane	0.014		546.91							118	691	363.7	
chloroethane	0.0799		546.91							27	8587	15944	
benzene	0.056		546.91							135	4178	1939.4	

(a) Based on concentrations at end of year 4; for detail see text

(b) Based on a deposition rate of 600 m/day on plant surfaces (McKone and Daniels, 1991)

(c) Interception fraction by plants (Copeland et al., 1994)

(d) Crop yield (Copeland et al., 1994)

(e) Weathering constant Ke replaced by a quantitative component that integrates evaporation from produce surfaces (McKone and Daniels, 1991); for details see text

(f) Growth period (Copeland et al., 1994)

(g) Concentration due to direct deposition

(h) Henry's Law constant (H) - Organic Carbon partition coefficient (Koc); value shown is an assumed triangular distribution based on values reported in MacKay et al. (1992) unless otherwise indicated (* uniform distribution based on 2 values)

(i) Uptake factor based on soil concentration

(j) Concentration due to root translocation or uptake-to be used to calculate intake of chemicals for potatoes

(k) Average concentration in and on leafy and vine crops

2) Concentration in plants (cont.)

	Crop Dep				**Root uptake (for Potatoes)**	**Deposition Only**	
	Cdepv (g) Leafy	Cdepv Vines	Koc (h)	UF2 (i)	Ctrans (j)	Cf leafy (k)	Cf Vines (k)
	ug/kg	ug/kg			ug/kg	ug/kg	ug/kg
1,1,2,2-tetrachloroethane	0.5316	0.0995	79	0.3663	0	0.002854	0.000534
bromodichloromethane	0.0583	0.0109	* 251	0.0822	0	0.00313	0.000586
1,1-dichloroethane	0.0115	0.0022	40	0.385	0	0.001734	0.000325
methylene chloride	0.0105	0.002	17	0.6457	0	0.003713	0.000695
1,1-dichloroethylene	0.0007	0.0001	* 150	0.142	0	8.32E-05	1.56E-05
vinyl chloride	0.0004	7E-05	32	0.3278	0	9.91E-06	1.86E-06
trichloroethylene	0.0222	0.0042	158	0.1905	0	0.000919	0.000172
1,2-dichloroethane	0.0283	0.0053	19	0.6482	0	4.78E-05	8.95E-06
1,1,2-trichloroethane	0.1333	0.0249	70	0.2859	0	0.000286	5.36E-05
chloroethane	0.003	0.0006	29	0.4136	0	3.68E-05	6.90E-06
benzene	0.025	0.0047	83	0.2567	0	0.00021	3.93E-05

Appendix B-III – I. Population Risk

2) Exposure through Inhalation

a) Inhalation variables associated with the receptors (extracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
IR (m3/d)	1.21	11	15.9	26.1	21.4
BW (kg)	6.28	13	26.8	50	61.5

IR was correlated to BW, r = 0.67

b) Exposure through Inhalation of Carcinogens

Chemicals	0-6m ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	5y-12y period 9-12 ug/kg-day	13y-20y period 13-16 ug/kg-day	13y-20y period 17-20 ug/kg-day	21+ period 21-25 ug/kg-day
1,1,2,2-tetrachloroethane	8.27E-04	3.63E-03	3.77E-03	4.14E-03	4.95E-03	7.45E-03	2.20E-03
bromodichloromethane	8.27E-03	3.63E-02	3.77E-02	4.14E-02	4.95E-02	7.45E-02	2.20E-02
1,1-dichloroethane	2.32E-02	1.02E-01	1.06E-01	1.38E-03	1.35E-01	2.11E-01	6.35E-02
methylene chloride	5.44E-02	2.39E-01	2.49E-01	2.83E-01	3.24E-01	4.94E-01	1.47E-01
1,1-dichloroethylene	1.77E-02	7.79E-02	8.05E-03	9.31E-03	1.04E-02	1.62E-02	4.79E-03
vinyl chloride	3.90E-03	1.71E-02	1.76E-02	2.00E-02	2.30E-02	3.48E-02	1.03E-02
trichloroethylene	6.38E-03	2.80E-02	3.02E-02	3.31E-02	3.78E-02	5.75E-02	1.71E-02
1,2-dichloroethane	2.60E-04	1.14E-03	1.21E-03	1.38E-03	1.58E-03	2.43E-03	7.33E-04
1,1,2-trichloroethane	3.31E-04	1.45E-03	1.48E-03	1.69E-03	1.94E-03	2.91E-03	8.79E-04
chloroethane	1.87E-03	8.20E-03	8.56E-03	9.65E-03	1.08E-02	1.70E-02	4.88E-03
I/O*** 0.75							
benzene	1.30E-03	5.69E-03	5.88E-03	6.69E-03	7.64E-03	1.17E-02	3.47E-03

* This RsC was calculated as for the LADDpot by considering receptor-specific inhalation rates and body weigths

** Calculated by assuming 23 m3/day and 70 kg

*** the Indoor/outdoor, I/O, ratio for long-term exposure was assumed to fit a triangular distribution ranging from 0.2 to 1(Hawkins, 1991) with a most likeliest value of 0.75 as determined for particulates (Hawley, 1985)

Appendix B-III – I. Population Risk

2) Exposure through Inhalation (cont.)

b) Exposure through Inhalation of Carcinogens

	21+ period 26-30 ug/kg-day	LADDpot ug/kg-day	Unit risk (ug/m3)-1	RsC 10E-6 ug/m3	Composite Receptor-based RsC 10E-6* (ug/kg-day)	Conventional RsC 10E-6** (ug/kg-day)	Ratio of Receptor-based to Conventional	ER	Risk
1,1,2,2-tetrachloroethane	2.07E-03	1.65E-03	5.80E-05	0.01724138	7.23E-03	5.67E-03	1.28	0.229	2.28572E-07
bromodichloromethane	2.07E-02	1.65E-02	2.00E-05	0.05	2.10E-02	1.64E-02	1.28	0.788	7.88193E-07
1,1-dichloroethane	5.59E-02	3.97E-02	1.60E-06	0.625	2.62E-01	2.05E-01	1.28	0.151	1.514E-07
methylene chloride	1.40E-01	1.10E-01	4.70E-07	2.12765957	8.93E-01	6.99E-01	1.28	0.123	1.23151E-07
1,1-dichloroethylene	4.47E-03	7.19E-03	5.00E-05	0.02	8.39E-03	6.57E-03	1.28	0.857	8.57111E-07
vinyl chloride	9.49E-03	7.75E-03	8.40E-05	0.01190476	4.99E-03	3.91E-03	1.28	1.551	1.55114E-06
trichloroethylene	1.62E-02	1.29E-02	2.00E-06	0.5	2.10E-01	1.64E-01	1.28	0.061	6.14535E-08
1,2-dichloroethane	6.71E-04	5.36E-04	2.60E-05	0.03846154	1.61E-02	1.26E-02	1.28	0.033	3.32174E-08
1,1,2-trichloroethane	8.38E-04	6.56E-04	1.60E-05	0.0625	2.62E-02	2.05E-02	1.28	0.025	2.50309E-08
chloroethane	4.64E-03	3.73E-03	1.80E-06	0.55555556	2.33E-01	1.83E-01	1.28	0.016	1.60208E-08
							total	3.84	3.835E-06
benzene	3.26E-03	2.60E-03	8.30E-06	0.12048193	5.05E-02	3.96E-02	1.28	0.051	5.14175E-08

3) Exposure through Dermal Contact with Soil - Carcinogens

a) Dermal exposure variables associated with the receptors (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	18y-20y	21+
SA (cm2) (assumed for active days) (b)	1746.66 (a)	2140	2380	3630	3400
BW (kg) (d)	6.28	13	26.8	50	61.5
SL (mg/cm2/day) (c)	0.52				

(a) No data available in IWA (1994); value derived from data in Finley et al., (1994) by using the triangular distributions for arms, hands, and legs, the distribution for surface area/body weight ratio for 0-2 infants and BW distribution from IWA (1994)

(b) Based on IWA (1994); SA was for active days

(c) Based on Finley et al. (1994)

(d) A correlation of 0.67 was assumed between BW and SA

b) Calculated exposure through dermal contact with soils * (Exposure through dermal contact with soils was assumed to occur for 7 days/week x 8 months/year)

Chemicals	Kow	Kh (a)	ABS (b)	0-6m (d) ug/kg-day	7m-4y ug/kg-day	period 5-8 ug/kg-day	period 9-12 ug/kg-day	period 13-16 ug/kg-day
1,1-dichloroethane	66	0.298	0.03	7.47E-04	4.42E-04	3.62E-04	1.78E-05	4.20E-04
vinyl chloride	14	3.21	0.03	1.43E-04	8.44E-05	6.95E-05	7.98E-05	8.46E-05
1,1-dichloroethylene	99	2.35	0.03	6.48E-04	3.84E-04	4.30E-05	3.72E-05	3.82E-05
bromodichloromethane	106	0.309	0.03	3.02E-04	1.79E-04	1.49E-04	1.65E-04	1.82E-04
1,1,2,2-tetrachloroethane	386	0.024	0.03	8.60E-06	5.09E-06	4.07E-06	4.45E-06	4.96E-06
methylene chloride	20	0.128	0.03	6.38E-04	3.77E-04	3.03E-04	3.44E-04	3.66E-04
trichloroethylene	531	0.304	0.03	6.49E-05	3.84E-05	3.18E-05	3.49E-05	3.70E-05
1,2-dichloroethane	34	0.035	0.03	5.26E-06	3.11E-06	2.54E-06	2.90E-06	3.08E-06
1,1,2-trichloroethane	146	0.028	0.03	1.97E-05	1.17E-05	1.08E-05	1.27E-05	1.34E-05
chloroethane	27	0.729	0.03	2.19E-05	1.30E-05	1.04E-05	1.17E-05	1.22E-05
benzene	152	0.222	0.03	5.04E-06	2.98E-06	2.37E-06	2.70E-06	2.86E-06

(a) Calculated based on Kh=P/(SxRxT) (see McKone, 1990)

(b) Fraction absorbed accross the skin; uptake in viable tissues reflective of a potential dose (see McKone, 1990); a maximal value of 0.03 was assumed based on the recommendation of McKone (1990)

(c) The slope factor for dermal exposure to carcinogens was assumed identical to the oral slope factor, when available, under the assumption of diffusion to venous blood and hepatic processing prior to distribution to other organs and tissues

(d) ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no contact with outdoor soil would occur during the first 3 months

3) Exposure through Dermal Contact with Soil - Carcinogens (cont.)

SA Variables for the 0-6m

	%	Total SA/kg	SA (cm2)
Arms	0.151	641	1747
Hands	0.0539		
Legs	0.229		

b) Calculated exposure through dermal contact with soils * (Exposure through dermal contact with soils was assumed to occur for 7 days/week x 8 months/year)

	period 17-20 ug/kg-day	period 21-25 ug/kg-day	period 26-30 ug/kg-day	LADDpot ug/kg-day	Oral slope factor (c) (ug/kg-day)E-1	RsD 10E-6 (ug/kg-day)	ER	Risk
1,1-dichloroethane	6.70E-04	2.42E-04	2.09E-04	9.53E-05	5.7E-06	1.75E-01	0.000543	5.43E-10
vinyl chloride	1.28E-04	4.58E-05	4.11E-05	2.13E-05	5.26E-04	1.90E-03	0.011222	1.12E-08
1,1-dichloroethylene	5.93E-05	2.14E-05	1.94E-05	2.39E-05	6.00E-04	1.67E-03	0.014349	1.43E-08
bromodichloromethane	2.73E-04	9.82E-05	8.95E-05	4.55E-05	6.20E-05	1.61E-02	0.002822	2.82E-09
1,1,2,2-tetrachloroethane	7.46E-06	2.54E-06	2.39E-06	1.24E-06	2.00E-04	5.00E-03	0.000248	2.48E-10
methylene chloride	5.58E-04	1.91E-04	1.82E-04	9.30E-05	7.50E-06	1.33E-01	0.000697	6.97E-10
trichloroethylene	5.63E-05	1.93E-05	1.83E-05	9.44E-06	6.00E-06	1.67E-01	5.67E-05	5.67E-11
1,2-dichloroethane	4.75E-06	1.67E-06	1.53E-06	7.84E-07	9.10E-05	1.10E-02	7.14E-05	7.14E-11
1,1,2-trichloroethane	1.97E-05	8.64E-06	7.17E-06	3.38E-06	5.70E-05	1.75E-02	0.000193	1.93E-10
chloroethane	1.92E-05	6.37E-06	6.05E-06	3.16E-06	1.30E-05	7.69E-02	4.11E-05	4.11E-11
							Total 0.030243	Total 3.02E-08
benzene	4.39E-06	1.49E-06	1.40E-06	7.28E-07	2.90E-05	3.45E-02	2.11E-05	2.11E-11

4) Exposure through Ingestion of Soil - Carcinogens

a) Variables associated with the receptors for soil ingestion (exctracted from IWA, 1994)

	0-6m	7m-4y	5y-12y	13y-20y	21+
Ing (mg soil/day) (a)	35	91	91	250	100
BW (kg) (b)	6.28	13	26.8	50	61.5

(a) Based on IWA (1994); Ing and BW were correlated at r= 0.67

(b) Based on IWA (1994)

b) Calculated exposure from soil ingestion * * Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year

Chemicals	ABS (a)	GI (b)	0-6m (c) ug/kg-day	7m-4y ug/kg-day	period 5-8 ug/kg-day	period 9-12 ug/kg-day	period 13-16 ug/kg-day	period 17-20 ug/kg-day
1,1-dichloroethane	1	1	9.59E-04	1.20E-03	8.88E-04	4.37E-05	1.85E-03	2.96E-03
vinyl chloride	1	1	1.83E-04	2.30E-04	1.70E-04	1.96E-04	3.74E-04	5.63E-04
1,1-dichloroethylene	1	1	8.33E-04	1.05E-03	1.05E-04	9.11E-05	1.69E-04	2.62E-04
bromodichloromethane	1	1	3.89E-04	4.88E-04	3.65E-04	4.05E-04	8.04E-04	1.21E-03
1,1,2,2-tetrachloroethane	1	1	1.10E-05	1.39E-05	9.97E-06	1.09E-05	2.19E-05	3.29E-05
methylene chloride	1	1	8.19E-04	1.03E-03	7.43E-04	8.43E-04	1.62E-03	2.47E-03
trichloroethylene	1	1	8.33E-05	1.05E-04	7.80E-05	8.54E-05	1.63E-04	2.49E-04
1,2-dichloroethane	1	1	6.76E-06	8.49E-06	6.23E-06	7.11E-06	1.36E-05	2.10E-05
1,1,2-trichloroethane	1	1	2.53E-05	3.18E-05	2.64E-05	3.12E-05	5.91E-05	8.68E-05
chloroethane	1	1	2.81E-05	3.53E-05	2.55E-05	2.88E-05	5.39E-05	8.49E-05
benzene	1	1	6.47E-06	8.13E-06	5.81E-06	6.61E-06	1.26E-05	1.94E-05

(a) Fraction desorbed from soil particles and bioavailable for uptake

(b) Fraction absorbed through the GI tract

(c) ADDpot for the 0-6m subgroup was estimated for the period 3-6m under the assumption that no ingestion of outdoor soil would occur during the first 3 months

Appendix B-III – I. Population Risk

4) Exposure through Ingestion of Soil - Carcinogens (cont.)

b) Calculated exposure from soil ingestion * * Exposure through soil ingestion was assumed to occur for 7 days/week x 8 months/year

period 21-25 ug/kg-day	period 26-30 ug/kg-day	LADDpot ug/kg-day	Oral slope factor (c) (ug/kg-day)E-1	RsD 10E-6 (ug/kg-day)		ER		Risk	
4.55E-04	3.95E-04	3.04E-04	5.70E-06	1.75E-01		0.001734		1.73E-09	1,1-dichloroethane
8.64E-05	7.75E-05	6.61E-05	5.26E-04	1.90E-03		0.0347536		3.48E-08	vinyl chloride
4.03E-05	3.65E-05	6.57E-05	6.00E-04	1.67E-03		0.0394043		3.94E-08	1,1-dichloroethylene
1.85E-04	1.69E-04	1.41E-04	6.20E-05	1.61E-02		0.0087474		8.75E-09	bromodichloromethane
4.78E-06	4.50E-06	3.85E-06	2.00E-04	5.00E-03		0.0007692		7.69E-10	1,1,2,2-tetrachloroethane
3.60E-04	3.44E-04	2.88E-04	7.50E-06	1.33E-01		0.0021609		2.16E-09	methylene chloride
3.64E-05	3.45E-05	2.92E-05	6.00E-06	1.67E-01		0.0001754		1.75E-10	trichloroethylene
3.14E-06	2.88E-06	2.43E-06	9.10E-05	1.10E-02		0.0002212		2.21E-10	1,2-dichloroethane
1.63E-05	1.35E-05	1.04E-05	5.70E-05	1.75E-02		0.0005913		5.91E-10	1,1,2-trichloroethane
1.20E-05	1.14E-05	9.80E-06	1.30E-05	7.69E-02		0.0001274		1.27E-10	chloroethane
					Total	0.088685	Total	8.87E-08	
2.81E-06	2.64E-06	2.26E-06	2.90E-05	3.45E-02		6.544E-05		6.54E-11	benzene

Appendix B-III – I. Population Risk

5) Exposure through Ingestion of Garden Produce - Carcinogens

a) Variables associated with the receptors for ingestion of garden produce (extracted from IWA, 1994)

				0-6m		7m-4y		5y-12y		13y-20y		21+
Ingestion (Kg/day) (a)	Leafy crops			1.54E-02		3.72E-02		4.75E-02		4.56E-02		4.28E-02
	Vine crops:	fruits	1.12E-01	1.54E-01	2.62E-01	3.11E-01	3.05E-01	3.76E-01	2.87E-01	3.82E-01	2.37E-01	3.51E-01
		vegetables	4.20E-02		4.90E-02		7.10E-02		9.50E-02		1.14E-01	
	Root crops			0.00E+00		6.10E-02		9.90E-02		7.40E-02		7.60E-02
BW (kg) (d)				6.28		13		26.8		50		61.5

(a) Based on IWA (1994); Ing for each produce category was correlated to BW (r = 0.67)

b) Calculated exposure from ingestion of garden produce* * Exposure through ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year

Chemicals	ABS (a)	GI (b)	UP (d)	0-6m (e) ug/kg-day	7m-4y ug/kg-day	5y-12y period 5-8 ug/kg-day	5y-12y period 9-12 ug/kg-day	13y-20y period 13-16 ug/kg-day	13y-20y period 17-20 ug/kg-day
1,1-dichloroethane	1	1	0.47	0.00E+00	3.46E-06	3.08E-06	4.02E-08	2.37E-06	3.70E-06
vinyl chloride	1	1		0.00E+00	1.98E-08	1.74E-08	1.98E-08	1.37E-08	2.08E-08
1,1-dichloroethylene	1	1		0.00E+00	1.66E-07	1.47E-08	1.70E-08	1.14E-08	1.79E-08
bromodichloromethane	1	1		0.00E+00	6.24E-06	8.62E-06	6.10E-06	4.40E-06	6.62E-06
1,1,2,2-tetrachloroethane	1	1		0.00E+00	5.69E-06	5.07E-06	5.56E-06	4.01E-06	6.03E-06
methylene chloride	1	1		0.00E+00	7.40E-06	6.63E-06	7.52E-06	5.20E-06	7.92E-06
trichloroethylene	1	1		0.00E+00	1.83E-06	1.69E-06	1.86E-06	1.28E-06	1.94E-06
1,2-dichloroethane	1	1		0.00E+00	9.53E-08	8.66E-08	9.88E-08	6.80E-08	1.05E-07
1,1,2-trichloroethane	1	1		0.00E+00	5.71E-07	5.00E-07	5.69E-07	3.93E-07	5.92E-07
chloroethane	1	1		0.00E+00	7.34E-08	6.58E-08	7.42E-08	5.01E-08	7.88E-08
benzene	1	1		0.00E+00	4.19E-07	3.72E-07	4.23E-07	2.91E-07	4.46E-07

(a) Fraction desorbed from plants and bioavailable for uptake

(b) Fraction absorbed through the GI tract

(c) The fraction of plant type homegrown (i.e., of local origin) was estimated from data reported in USEPA (1990): leafy: 0.7, root crops: 0.2, vines: 0.5.

(d) Fraction of the total mass of ingested fruits and vegetables that consist of unprotected produce (McKone and Daniels, 1991)

(e) The 0-6m subgroup was assumed to be breastfed and/or formula-fed

5) Exposure through Ingestion of Garden Produce - Carcinogens (cont.)

b) Calculated exposure from ingestion of garden produce* * Exposure through ingestion of garden produce was assumed to occur for 7 days/week x 8 months/year

21+ period 21-25 ug/kg-day	21+ period 26-30 ug/kg-day	LADDpot ug/kg-day	Oral slope factor (c) (ug/kg-day)E-1	RsD 10E-6 (ug/kg-day)	ER	Risk	
1.26E-06	1.11E-06	5.82E-07	5.70E-06	1.75E-01	3.32E-06	3.32E-12	1,1-dichloroethane
6.93E-09	6.41E-09	4.05E-09	5.26E-04	1.90E-03	2.128E-06	2.13E-12	vinyl chloride
5.97E-09	5.58E-09	8.45E-09	6.00E-04	1.67E-03	5.068E-06	5.07E-12	1,1-dichloroethylene
2.21E-06	2.08E-06	1.40E-06	6.20E-05	1.61E-02	8.677E-05	8.68E-11	bromodichloromethane
2.01E-06	1.90E-06	1.17E-06	2.00E-04	5.00E-03	0.0002339	2.34E-10	1,1,2,2-tetrachloroethane
2.66E-06	2.54E-06	1.54E-06	7.50E-06	1.33E-01	1.156E-05	1.16E-11	methylene chloride
6.54E-07	6.20E-07	3.82E-07	6.00E-06	1.67E-01	2.29E-06	2.29E-12	trichloroethylene
3.58E-08	3.28E-08	2.02E-08	9.10E-05	1.10E-02	1.838E-06	1.84E-12	1,2-dichloroethane
2.02E-07	1.93E-07	1.17E-07	5.70E-05	1.75E-02	6.65E-06	6.65E-12	1,1,2-trichloroethane
2.56E-08	2.43E-08	1.51E-08	1.30E-05	7.69E-02	1.968E-07	1.97E-13	chloroethane
					Total 0.000354	Total 3.54E-10	
1.50E-07	1.40E-07	8.66E-08	2.90E-05	3.45E-02	2.51E-06	2.51E-12	benzene

Summary Tables for All Pathways

Risks Associated with Exposure to Carcinogenic Chemicals at the Maximum GLC - Deterministic Approach

	Inhalation		Ingestion				Dermal		Total risk per chemical (a)
Chemical	Vapours		Soil		Garden produce		Soil		
Chlorinated Compounds									
1,1,2,2-tetrachloroethane	2.2857E-07	99.46%	7.69E-10	0.33%	2.339E-10	0.10%	2.5E-10	0.11%	2.29823E-07
bromodichloromethane	7.8819E-07	98.54%	8.75E-09	1.09%	8.677E-11	0.01%	2.8E-09	0.35%	7.99849E-07
1,1-dichloroethane	1.512E-07	98.51%	1.73E-09	1.13%	3.316E-12	0.00%	5.4E-10	0.35%	1.535E-07
methylene chloride	1.2315E-07	97.72%	2.16E-09	1.71%	1.156E-11	0.01%	7E-10	0.55%	1.26021E-07
1,1-dichloroethylene	8.5711E-07	94.10%	3.94E-08	4.33%	5.068E-12	0.00%	1.4E-08	1.58%	9.1087E-07
vinyl chloride	1.5511E-06	97.12%	3.48E-08	2.18%	2.128E-12	0.00%	1.1E-08	0.70%	1.59712E-06
trichloroethylene	6.1454E-08	99.62%	1.75E-10	0.28%	2.29E-12	0.00%	5.7E-11	0.09%	6.16878E-08
1,2-dichloroethane	3.3217E-08	99.12%	2.21E-10	0.66%	1.838E-12	0.01%	7.1E-11	0.21%	3.35118E-08
1,1,2-trichloroethane	2.5031E-08	96.94%	5.91E-10	2.29%	6.65E-12	0.03%	1.9E-10	0.75%	2.58216E-08
chloroethane	1.6021E-08	98.96%	1.27E-10	0.79%	1.968E-13	0.00%	4.1E-11	0.25%	1.61895E-08
subtotal	3.835E-06		Sub-total 8.87E-08		sub-total 3.54E-10				
Total inhalation risk (b)	3.84E-06	96.98%	Total ingestion risk		8.9E-08	2.25%	Total dermal 3.0E-08	0.76%	3.954E-06
Others									
benzene			6.54E-11		2.51E-12				
Total inhalation risk	5.14E-08	99.83%	Total ingestion risk		6.8E-11	0.13%	Total dermal 2.1E-11	0.04%	5.15066E-08

(a) The conceptual assumption of additivitiy across different routes may not be valid for this compound due to differences in route-specific target organs and end-points

TECHNICAL REPORT – PART "C"

ASSESSMENT OF THE POTENTIAL ECOLOGICAL RISKS ASSOCIATED WITH CHEMICAL EMISSIONS FROM MODERN MUNICIPAL SOLID WASTE INCINERATORS OR LANDFILLS IN ONTARIO - AQUATIC ASPECTS

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Preface

This assessment of potential ecological effects arising from either landfilling or incineration has been done as part of a larger baseline study by the Ministry (see Reports A, B, and D). Ministry, national or U.S. guidelines for ecological hazard/risk assessment have been used for this assessment. This study uses extensive data collected for an environmental assessment for a proposed landfill. However, the assessment is predictive and hypothetical in that data for landfilling are composites from across Ontario. Incineration data are taken from a single representative facility, but use the proposed landfill site characteristics.

Many assumptions have been made for this assessment. In addition, the assessment is not straightforward since many of the gaseous emissions from incineration will not be deposited in the local area but will become part of the province-wide load of contaminants in the airshed. Contaminant emissions from incineration cease with the closing of the facility, whereas landfills continue to emit contaminants in gases and leachate for many years following closure. Potential problems with landfills tend to be more localized than for incineration – in other words, the comparison is partly biased since landfilling is not the beneficiary of the dilution provided to incineration. The critical question is whether a meaningful comparison may be made given the uncertainties and assumptions which have been used in this assessment.

It is essential to remember that this assessment is not exhaustive. It is based on available data to which scientific judgement has been applied. Calculations include simplifying assumptions and default values for many variables.

Summary

A preliminary comparative ecological risk assessment was undertaken of the risks to aquatic life from two options for municipal solid waste disposal – incineration or landfilling. We have assumed that best management practices and best available facilities will be used in both incineration and landfilling practices. Nevertheless, these disposal methods emit different types and amounts of chemicals and have the potential for different risks to aquatic life. The assessment procedure loosely followed the Ecological Risk Assessment Framework for Contaminated Sites in Canada (CCME 1995) as well as aquatic hazard assessment approaches. The comparison relied heavily on data from existing risk assessments, especially that of the Interim Waste Authority assessment for a proposed landfill site in Durham (IWA 1994a). The IWA also provided a wealth of geological, hydrological and meterological background on the Duffins Creek watershed for the "hypothetical site". Emission data from many southern Ontario waste sites were also used in the comparison.

During the twenty-year life span of a disposal facility, and after closure in the case of landfill, contaminants may reach surface water from several pathways: direct deposition of airborne contaminants to surface water, runoff of contaminants deposited to the watershed and erosion of contaminated soil contributing to aquatic sediments. In addition, landfills may also discharge treated leachate to surface water, and groundwater may become contaminated and discharge to surface waters.

Lists of chemicals in emissions came from monitoring data for existing facilities. Data from the KMS Peel Inc.,incinerator were used without any initial screening out (i.e. exclusion) of chemicals. For landfilling, extensive data were available from the IWA 1994d report which compiled hundreds of chemicals. In that report, chemicals were subsequently screened down to a shorter list of about 60 chemicals in either air, leachate or groundwater, and this list became the starting point for the present analysis.

The evaluation consisted of an exposure assessment – prediction of contaminant concentrations in surface waters and sediments – and hazard assessment – measure of toxicity – in this case, environmental quality criteria. The risk/hazard characterization is based on the exposure/hazard ratio.

The estimation of chemical concentrations in surface waters potentially arising from landfilling or incineration practices began with the estimation of ground-level concentrations in air which were modelled from air emissions data. The mass of contaminants deposited directly and indirectly to surface water was calculated. Then waterborne concentrations were derived using stream flow rates. In the case of landfilling, leachate concentrations and groundwater contributions to surface water also were considered. Leachate concentrations were decreased by factors to account for on-site pretreatment and final sewage treatment and dilution in the main sewage treatment plant.

The most difficult pathway to quantify was runoff from land to water. Many of the chemicals in air emissions from landfilling are volatile, quickly evaporating from soils. Thus it was assumed that runoff of contaminants from landfilling was negligible. For incineration, it was arbitrarily assumed that runoff contributed 10-fold greater mass of chemicals than direct deposition. Based on the data from Duffins Cr. watershed, this equated to the movement of about 40% of deposited chemicals to surface water annually. Similar assumptions were made in evaluating sediment. It was assumed that sediment quality would eventually become identical to surrounding soil, i.e. some contamination from incineration, but negligible from landfilling.

A screening level assessment was undertaken by comparing expected environmental concentrations (EEC) to hazard estimates based on Provincial Water Quality Objectives (OMEE 1994) or Provincial Sediment Quality Guidelines (OMEE 1993). Where these criteria did not exist, criteria from other agencies were used, or surrogate criteria derived. These criteria represent safe concentrations and account for the potential for bioaccumulation but generally do not consider mixtures or interactions between chemicals.

In the case of incineration, only **mercury,** and **dioxins/furans** were not screened out. Further evaluation, made use of more realistic assumptions and making use of chemical mobility characteristics. Dioxins/furans, for example, are primarily retained in soils. This assessment indicated that there is little likelihood of mercury and dioxins/furans reaching surface water in excess of their respective water quality criteria. Aquatic biota would therefore be protected. Wildlife which subsist largely on fish were not evaluated in this assessment due to the lack of tissue residue criteria for wildlife protection. In addition, there is much public concern regarding mercury and dioxins/furans and these chemicals should be further evaluated.

In the case of landfilling, chemicals remaining after the initial screen were **toluene** and three alkanes: **octane**, **nonane**, and **3-methylheptane**. For the alkanes, surrogate criteria were derived based on chemical properties and were probably overly conservative. In a further assessment, equilibrium partitioning was incorporated through the use of a level I fugacity model. All three alkanes are highly volatile. Equilibrium partitioning would be almost completely to air. In addition, these chemicals also degrade in the environment with half-lives from 3 to 17 weeks. Detection of these chemicals in water arising from landfilling operations is highly unlikely. Toluene initially exceeded its PWQO by about 2 fold. However, a partitioning model indicated that almost all of the toluene would remain in air, with little impact on water quality. Leachate resulting from landfilling activities would be toxic until treated. However, leachate treatment in sewage treatment plants can effectively mitigate toxicity and this assessment assumes effective leachate treatment. Groundwater contamination in landfilling activities also may occur. However the soil, hydrogeological features and design features of the IWA site were selected such that the potential for contamination of groundwater was remote. Reasonable worst-case groundwater concentrations never exceeded PWQOs.

Chemical concentrations in soils near an incinerator were predicted to change little over the 20 years of accumulation even in areas of highest deposition (see report on terrestrial effects). Sediments in the vicinity of an incinerator, in this scenario, could show elevated concentrations of mercury and manganese. Also, chlorinated dioxins and furans exceeded the proposed Canadian Sediment Quality Guidelines (CCME 1994) by a factor of 0.6. However, these concentrations would not normally require remediation.

In summary, both methods of waste disposal result in chemical emissions of different kinds, but local ecological impact is unlikely. The comparison relied heavily on monitoring data from existing sites, but made many assumptions regarding the behaviour of these emissions. The comparison is less than exact since wastes are handled differently. Incineration wastes are widely dispersed and may add to long range loads whereas landfilling wastes tend to be localized but with some contribution to greenhouse gases (see Report D for further details) and acid gases. On the basis of this comparative risk assessment, as it pertains to chemical effects on aquatic life, neither disposal method is expected to have demonstrable local effect. Chemicals associated with incineration, mercury and dioxins and furans, may be of greater relative concern due to toxicity and persistence, than are chemical emissions arising from landfilling.

C-1.0 INTRODUCTION

The purpose of this report is to compare relative risks to the aquatic environment from two options for disposal of municipal solid waste: landfilling versus incineration. This comparison uses *A Framework for Ecological Risk Assessment* (CCME 1995) which is similar to USEPA methodology (USEPA 1992) as well as current thinking in ecological risk assessment (eg. Suter 1993).

This risk assessment uses a simple screening-level approach which is similar to earlier hazard assessment techniques (Cairns et al 1978, Suter 1990). More advanced qualitative, or semi-quantitative deterministic approaches, or probabilistic approaches, which make use of ranges of emissions, environmental concentrations and organism exposures to derive risk probabilities, are beyond the scope of this assessment.

The relative risk scenario for landfilling/incineration is hypothetical but relies on composite data from actual sites in southern Ontario. For landfilling, the scenario is based on reports of the Interim Waste Authority on the proposed Durham landfill site EE11 (IWA 1994a) in the Duffins Creek watershed. These reports describe the physical setting and projected chemical emission rates from the site.

Many assumptions and approximations have been made during this analysis and are documented here or referenced. The Durham site was also used as the setting for the hypothetical incinerator. However, actual emission data from the KMS Peel Inc.,facility were used (described elsewhere in this report). In some cases worst-case emissions were assumed in characterizing exposures.

C-2.0 ECOLOGICAL RISK ASSESSMENT PLANNING

C-2.1. Objective

The objective was to determine *relative* risks from two hypothetical facilities handling similar amounts of waste for a similar time period but in two very different ways. Incineration burns waste, perhaps recovering energy, and rapidly oxidizing many, but not all potential organic contaminants, while producing a residue high in inorganic elements such as metals. Waste products, including many organic chemicals, are expelled into the atmosphere where they may precipitate locally as particulates. Volatile compounds may not precipitate in the area. Only a small percentage, < 15%, of the waste products emitted from the stack will precipitate in the immediate area (400 km^2). The remainder can contribute to the province-wide load of contaminants in the airshed. The potential impacts of incineration on the environment would be immediate and would end with the closure of the facility.

In contrast, air emissions from landfills are primarily organic chemicals which may be burned off in flares. Many persistent chemicals collect in site leachate which must then be treated.

Landfill emissions reach a peak late in the operating life of the facility and decline only slowly following closure.

In short, the objective was to establish the potential for acute toxicity, chronic toxicity or the discharge of persistent bioaccumulative contaminants from either landfill or incineration.

C-2.2 Overview of the Durham site

The site is located in Southern Ontario in a populated area near the ecologically-important Rouge River. Details of the watershed are included in the IWA report (IWA 1994c) which indicate that the area is drained by West Duffins and Ganatsekiagon Creeks. These both flow into Duffins Creek above Highway 2 and then into Lake Ontario. Groundwater flow is well characterized. Several additional sources of contaminants impinge upon the Durham site. For example, there is a sewage treatment plant upstream at Stouffville and many non-point sources of pollutants to the surface water (G. Bowen pers. comm.) but these were excluded from the comparison because they are site-specific.

C-2.2.1 Leachate

Leachate treatment design consisted of collection and pretreatment in an on-site sewage treatment plant providing tertiary treatment, transfer by pipeline to the local sanitary sewer system with subsequent dilution and treatment in a municipal sewage treatment plant before discharge to surface water.

C-2.2.2 Stormwater

Site stormwater would be held in a retention pond for settling of particulates before discharge into a small tributary of Ganatsekiagon Creek (IWA 1994a, p1-2). Stormwater was not considered further in the comparison because the composition was not characterized in the IWA report. This aspect should be reviewed more fully in any risk management decision.

C-2.2.3 Groundwater

Groundwater flow has been modelled. It was estimated that groundwater contributions to stream flow (= base flow) were 46% and 63% of the total flow in West Duffins and Ganatsekiagon Creek respectively, with contributions to Urfé creek about the same as to Ganatsekiagon (IWA 1994a p.4-15).

C-2.2.4 Air Emissions

Meterological conditions at the site were based on data from Stouffville Airport (wind speed and direction and temperature) and Buttonville Airport (cloud cover) (IWA 1994a, p.4-4). As of

June 1994, a meterological site was established at EE11, but data were not available. Using data from the adjacent Brock West Landfill, the estimated landfill gas collection efficiency is 60% with a 98% destruction efficiency for non-methane organic compounds in the energy recovery plant (IWA 1994a p4-7) with subsequent production of nitrogen oxides.

C-2.3 Identification of Valued Ecosystem Components

Identification of valued ecosystem components is an essential part of a site-specific ecological risk assessment (CCME 1994). However, this comparative risk assessment criteria has been undertaken at the screening level. As such, it uses water quality and other environmental criteria which have been developed to protect the most sensitive aquatic species for an indefinite exposure. This is a conservative approach, extending protection to more than just the valued ecosystem components. It is also consistent with Ministry policy (OMEE 1994).

C-2.4 Conceptual model.

A conceptual model of the site, given in Figure 1, indicates the fate pathways of contaminants from both landfill and incinerator. Some pathways will be more relevant to one disposal method than to the other. For example, aerial deposition will be more important in incineration than landfilling. Actual chemicals being deposited via a given pathway will also vary between scenarios (see Exposure Assessment below).

To facilitate the comparative risk assessment it was decided that the site would receive up to 2000 tonnes/d (intensity) of either landfill or incineration for a twenty-year life (duration). Lists of contaminants emitted from incineration of municipal solid waste were compiled for the KMS Peel Inc.,incineration facility. The ground level concentration (GLC) of particulates was modelled using Stouffville airport meterological data. Wet deposition was also estimated for the area (see below). This resulted in a dilution factor from stack gas emission rates to ground level concentration. These input GLCs were used in the screening and final assessments below.

C-3.0 SCREENING ASSESSMENT

This preliminary assessment follows the scheme to the right. It compares the expected environmental concentration (EEC) with EQC. Where the EEC is lower than the corresponding criteria, i.e. risk ratio <1, there is no risk on a chemical-by-chemical basis. For the purposes of this screening assessment it is assumed that the safety/uncertainty factors used in deriving the criteria will also provide protection against chemical mixtures and interactions. That is, non-additivity is assumed.

Waterborne chemical concentrations were modelled or predicted for the full list of chemicals emitted from the KMS Peel Inc.,incinerator and shorter, pre-screened, lists of chemicals emitted from landfills.

The screening consisted of calculating the risk ratio: EEC/EQC. If the estimated concentration did not exceed the criterion, i.e. the ratio was <1, then the chemical was screened out as contributing no risk of adverse effects on meaningful population endpoints such as survival, growth and reproduction. No specific provisions were made for bioaccumulative substances other than the use of a larger uncertainty factor in deriving the EQC. As with other complex effluents, compliance was determined on a chemical-by-chemical basis with the assumption of non-additivity at concentrations at or below the PWQO.

C-3.1 Hazard Identification - Selection of Chemicals Emitted from Landfilling or Incineration Practices

Hazard identification looks at the types and amounts of chemicals emitted. For identification of chemicals arising from incineration, monitoring data from the KMS Peel Inc.,Incineration facility were used (the incinerator facility is described in more detail in the Human Health Risk Assessment).

For landfilling, the chemical listing and preliminary screening were done by consultants for the Interim Waste Authority for site EE11. Three separate source tables were used: vent/flare emissions to air (IWA Table QD.19), leachate concentrations (IWA Table QD.21), and groundwater concentrations (IWA Table QD.28) (all in IWA 1994d). These separate tables were then combined giving 90 chemicals.

For vent/flare emissions from landfills, IWA examined southern Ontario and U.S. data and chose vent chemicals and concentrations based on the median concentrations in vents at the Brittania Road (Mississauga, ON) landfill. That site has no gas collection system and no flares and is thus a worst case. Chemicals detected at other landfills were also included based on maximum concentrations since median data were unavailable. The final vent unflared concentrations are presented in Table QD.19 (IWA 1994d pQD-76).

IWA calculated two scenarios for groundwater: an expected concentration, and a reasonable worst case. Their initial screening was based on two considerations: i) elimination of all compounds which would not migrate through the soil to any extent based on an octanol: water partition coefficient (K_{oc}) > 5000, ii) chemicals seldom detected in leachates (95% or more of samples were non-detectable for the chemical). From a short list of 58 chemicals, groundwater concentrations in the aquifer underlying site EE11 were estimated by IWA using the POLLUTE model.

C-3.2 Exposure Assessment - Estimation of Air, Soil, Sediment, Runoff, Leachate and Groundwater Contributions to Chemical Concentrations in Surface Water

Air concentration. The ground level concentration was given as:

GLC = E-rate * X/Q 1)
(CAPCOA 1993)

where - GLC is the ground level concentration (ug/m^3)
- E-rate is the pollutant emission rate (g/s)
- X/Q is the dilution factor provided by dispersion modelling ug/m^3 per g/s of emissions

GLCs were provided by Ministry air modelling staff as either annual averages or daily averages for a 20 x 20 km land area. For this comparison, the maximum annual average concentration was used. For landfilling, ground level concentrations were derived for the worst case emissions (years 17-21) by modelling the annual average GLC for vinyl chloride for that time period, then taking the average GLC over a 400 km^2 grid. GLCs for other chemicals were calculated using the ratio of the vent concentration to the calculated GLC for vinyl chloride. Vinyl chloride was selected because of its high concentration and toxic potential, but any of the compounds could have been used since the relationships are essentially linear.

Soil concentration. Although not used directly, it was necessary to calculate soil concentrations resulting from the incineration practices since it was assumed that sediment concentrations would eventually equal soil concentrations after 20 years of deposition. This approach was only used for incineration since it was also assumed that volatile organic compounds (VOCs) from landfilling would not accumulate in sediments.

soil concentration = deposition * X / (K_s * SD * BD * T_t) 2)
(CAPCOA 1993)

where - dep is deposition (dry) from equ. 3 below
- X is a loss function
- K_s the soil elimination constant
- S the mixing depth (5 cm)
- BD the bulk density
- T_t the total accumulation time (20 years)

For initial screening we have assumed no loss, thus X and K_s are excluded from equation 2.

Sediment concentrations for incineration, were set equal to the soil concentration after 20 years of accumulation (see terrestrial assessment for more detail). This is conservative since final sediment quality would be composed of cleaner upstream sediment and other eroded bank material. As well, contaminants could be removed over time through dissolution, or degraded (degradation half lives were not considered). For landfilling, due to high rates of volatilization from runoff, surface water and soil, the potential for sediment accumulation was considered negligible.

Runoff. The contribution to surface water from runoff is a function of soil concentration, mobility of the contaminant, and amount and type of precipitation. The runoff pathway is not well-considered by most risk assessments. This includes the Massachusetts approach used in the soil decommissioning guidelines (A. Kuja pers comm.) and the CAPCOA model (CAPCOA 1993). The Laidlaw Environmental Services Ltd. Lambton County Risk Assessment for Incineration (Clement 1991) estimated runoff contribution to a drainage ditch as two or four times the concentration arising from direct deposition. For the screening level assessment, given the surface water area below, if all deposited contaminants were carried to surface water the runoff component would be 24-fold the direct deposition (96% land / 4% water). For incineration we have assumed a factor of ten or about 40% of material deposited reaching surface water from runoff. For landfilling, runoff was considered negligible due to the high volatility of most chemicals, and the presence of a stormwater retention pond for the settling of particulates.

Leachate concentrations from landfilling were taken from the IWA report. Leachate from landfilling was well-characterized by the IWA. Leachate chemical concentrations were reduced by sewage treatment and dilution. Reduction values due to sewage treatment were taken from the literature where possible otherwise a default value of 50% was used. There was also considerable dilution in the sewage treatment plant (see below). Incineration produces both fly ash and bottom ash. Typically bottom ash is considered non-hazardous and is landfilled with municipal solid waste. Fly ash which fails leachate testing is considered hazardous, requiring special disposal in a hazardous waste landfill. Treatment efficiency values for incinerator ash were not available. However, for screening purposes it is assumed that metals and refractory organic chemicals from incinerator ash would be immobilized by organic carbon and other landfill constituents such that leachate would contribute little to surface water concentrations.

Groundwater concentrations from landfilling were taken from the IWA report. The report provided both expected concentrations and reasonable-worst-case concentrations. The latter were used, to be more conservative. It was assumed that landfilled ash from incineration would not contribute to groundwater contamination due to the immobility of metals in soils, especially in the clay liners of well-designed landfills.

Waterborne concentrations of chemicals reaching surface water from either incineration or landfilling were calculated as the sum of direct deposition to water, runoff from the watershed, groundwater contributions and landfill leachate discharges. Pathways differed depending on source (Fig C-2.1). In the initial screening, chemical degradation or loss was assumed to be nil. Actual site data were used to obtain the fraction of surface water to total watershed area as well as water flow rates. To facilitate comparisons, all deposition and flow rates were expressed on a m^2 basis. Wet deposition was calculated as 30.8 $ug/m^2 \cdot d$. For landfilling, wet deposition of organic chemicals does constitute a potential input and might be modelled as a function of the Henry's Law constant (partition coefficient between water and air), air column chemical concentrations, contact time between air and raindrop, and precipitation duration. This modelling was not done because it is believed to be a relatively minor pathway to surface water for these chemicals, and is covered in part under dry deposition calculation.

$$\text{Deposition (dry, in } ug/m^2 \cdot d) = GLC * \text{dep-rate} * 86\,400 \qquad 3)$$

(CAPCOA 1993)

where - GLC is from equation 1) above

- dep-rate is the deposition rate in m/s. The default value was 0.02 m/s for particulates, and 0.0065 m/s (McKone and Daniels 1991) for volatile organics
- 86 400 converts seconds to days

These procedures are especially conservative for landfilling, since many of the chemicals would not deposit at all, due to high volatility.

water concentration = (((dry dep + wet dep) * SA) + runoff)/ Q 4)

(after Clement 1991 c-51)

where

- dry deposition from equation 3) above in ug/m^2·d
- wet deposition is 6.51 ug/m^2·d per g/s of emission from incineration (R. Bloxam pers. comm.)
- SA is the fraction of water surface area = 0.04
- runoff = 10 (dry dep + wet dep)
- Q is base flow rate of 0.4 L/m^2·d

For incineration, all the inputs were summed and a final concentration calculated. For landfilling, some chemicals were unique for a given pathway of air, leachate and groundwater sources. Some chemicals were common to two or all three pathways. While it is desirable to sum those inputs, in practice, concentrations were so low that the sum was well below the screening criteria, or, one source (leachate) was so concentrated that summing of the other sources made little difference to the EEC. Summation might have been significant for two chemicals. For nitrate, groundwater was 14% that of leachate. The next highest percentage was for 1,1-dichloroethane, at 9%. Concentrations were still so low that both chemicals were screened out.

C-3.3 Hazard Assessment

Hazard or "effects" assessment consists of deriving some measure of toxicity of the chemicals emitted from a facility. In this assessment, environmental quality criteria were used. These were Provincial Water Quality Objectives (OMEE 1994) or Provincial Sediment Quality Guidelines (OMEE 1993), Canadian Water Quality Guidelines (CCREM 1987), or some surrogate. PWQO is lowest water concentration that caused significant toxic effect in laboratory tests, divided by safety factor of 10 or more. PSQG are based on field surveys of biota and sediment contaminants or on PWQO converted to a sediment-based concentration using a partition coefficient. Where criteria did not exist, surrogates were derived by taking the lowest toxicity data available and dividing it by a safety/uncertainty factor of 20 in the case of metals. For organics, 200 was used, or in the case of a bioaccumulative substance (log K_{ow}>4), 2000. If actual toxicity data did not exist then toxicity values were estimated from physical-chemical parameters using quantitative structure/activity relationships (QSAR) available on the ASTER database (Assessment Tools for Evaluation of Risk 1995).

C-3.4 Risk/Hazard Characterization

In the risk characterization process, once the chemicals are identified and their EECs calculated, they are compared with hazard estimates to determine overall risk. Hazard estimates are often in the form of probable "no effect" concentrations. In the present case however we used EQC, usually PWQO, since they are more easily found for many of the chemicals and, in addition, have safety factors applied. Thus the risk characterization was conservative.

A risk ratio for individual chemicals was calculated as:

$$\text{risk} = \text{EEC} / \text{EQC} \qquad 5)$$

where
- EEC is the expected environmental concentration (ug/L or for sediment, ug/g)
- EQC the chemical-specific environmental quality criterion (ug/L or for sediment, ug/g)

Chemicals with ratios < 1 were screened out as unlikely to cause aquatic environmental effects based on the conservative assumptions and worst-case exposures and hazards that were used in the screening. No further consideration or action is necessary for these chemicals.

C-3.4.1 Risk Characterization: Incineration

Surface water. Of a total of 82 chemicals, seven did not have criteria or surrogate criteria (Table C-3.1). Two of these, calcium and magnesium were elements which are not considered toxic, are normal constituents of natural waters. A third, silicon, is essentially insoluble.

Three of the chemicals without EQC were acid gases (sulphur dioxide, hydrogen chloride and nitrogen oxides). The risk of prolonged gas emissions would be due to chronic effects caused by gradual acidification of the environment (e.g. loss of alkalinity, metal mobilization) rather than direct acute toxicity. The gases are not bioaccumulative. Water pH changes and aquatic effects depend on the alkalinity and buffering capacity of the receiver. The Great Lakes do not suffer acid rain damage, but the soft water lakes on the Canadian Shield are sensitive. Generally, air emissions would adsorb to terrestrial vegetation and soil, so the direct runoff to surface waters would be minimal rather than the 40% of total deposition that was assumed.

Inputs from runoff might eventually increase if the terrestrial environment were acidified and damaged. Acid gas emissions do have environmental impacts, so the risk is properly assessed in the context of environmental loadings of acid gases from all sources. It is likely that incineration of municipal waste would be small in comparison. For example, loadings of wet and dry sulphur to the watershed would be about 0.01 kg S per hectare per year, or < 0.1% of the 20 kg/ha·a guideline to protect acid-sensitive lakes (reference for 20 kg/ha·a). Total sulphur deposition would be about 0.03 $g/m^2 \cdot a$. Compared with the 1983 deposition rate for this area of

1.5 g/m^2·a (Neary and Dillon 1988) this would represent about a 2% increase. Hydrogen chloride at about the same concentration and even nitrogen oxides at about 20-fold higher deposition would not cause an effect on surface water pH in the area since it is generally well buffered.

For organic compounds, water concentrations in the low parts per trillion (ppt) range were estimated for 1,2,3,4-tetrahydronaphthalene (= tetralin, MW 132) and *o*-terphenyl (MW 230). Organic compounds of moderate molecular weight, such as these, generally have guidelines in the low ppb range, assuming that they are not persistent or bioaccumulative. In this respect the estimated low ppt levels probably would not cause aquatic environmental effects. Nevertheless, PWQO development should be undertaken for these compounds. At a minimum, their potential persistence and bioaccumulation should be characterized.

Chemicals with ratios > 1 may cause environmental impacts, so further consideration is necessary. Ratios above one were calculated for two substances or substance groups: mercury, and combined total toxicity equivalents for dioxins/furans.

Mercury (hazard ratio ~ 10). Most metals remain in the incinerator ash, but mercury is volatile, so proportionately larger stack emissions occur. The estimated total mercury concentration in surface water (2 ug/L) exceeded the PWQO (0.2 ug/L). The two are not directly comparable however since the PWQO is for methylmercury (IJC 1976), operationally defined as the dissolved fraction (0.45 um filter pore size). This hazard ratio was thus re-evaluted using more realistic assumptions and the specific properties of mercury. The stack gas concentration was reduced by a factor of 0.135 to comply with the recent MOEE A-7 incineration guidelines. GLC concentrations were averaged over a larger area of the watershed instead of using the maximum. Runoff was reduced to 15% which represents a consensus value for release from upland areas (G. Mierle pers. com.). The methylmercury fraction can be quite variable but was set at 50% for this assessment. After these revisions the EEC for mercury was well below the EQC (Table C-3.2) indicating that aquatic life will be adequately protected. Nevertheless, the EEC is considerably above the background concentration of <1 ng/L in oxic waters (see Wiener and Spry 1996).

Environmentally, mercury is still a contaminant of concern. In 1990, fish consumption restrictions were recommended for 90% of walleye, 80% of smallmouth bass and 60% of lake trout populations in over 1200 Ontario lakes due to the presence of mercury (OMEE, OMNR 1990). In herring gulls, a sentinel species on the Great Lakes, mercury levels have declined or remained constant from 1972 to 1992 as direct industrial inputs have been curtailed. There are still inputs to the Great Lakes from municipal waste processes and resuspension from sediments (Weseloh et al. 1996). Borgmann and Whittle (1992) indicated that mercury levels in some Lake Ontario fish have declined 50% between 1977 and 1988 and estimated a half-life for mercury in Lake Ontario fish of about 11 years.

Mercury is thus a known persistent, bioaccumulative and toxic substance. While levels in biota in many areas are declining, there are still mercury-related restrictions on Ontario sport fish consumption. A screening assessment is insufficient to make firm conclusions about the aquatic environmental impacts of mercury from incinerators. A more detailed assessment may provide better estimate actual risks. This should include an assessment based on estimated fish residues. In addition, the effects on fish-eating wildlife were not considered, largely due to the lack of tissue residue guidelines (these are under development nationally). The US Water Quality Guidance for the Great Lakes recently promulgated a water quality criteria to protect wildlife of 1.3 ng/L which is lower than the revised EEC here. This further supports additional assessment. We also recommend that the air emissions should be compared to other Ontario sources (for example, natural bedrock erosion and gold-mining activities). The recommendation is to limit mercury inputs as much as possible.

Dioxins and furans (total TEQs, hazard ratio ~ 3). The estimated total toxic equivalents (TEQs) of dioxins and furans in surface water (1 pg/L) exceeded the Proposed Canadian Water Quality Guideline (0.4 pg/L) by a small factor. Applying the MOEE A-7 incineration guidelines and allowing for greater watershed dispersion would reduce the EEC by 0.52 and bring it below the criterion even assuming a 40% runoff (Table C-3.2). However, these compounds are hydrophobic, so the contribution to water from land runoff is a considerable over-estimate. In fact, a level III fugacity model indicated that 92% of 2,3,7,8-TCDD discharged to air would be retained in soils. A more realistic value for this pathway would further reduce the EEC. Nevertheless, municipal incinerators are a known source of dioxins and furans, and some members of this group are classed as persistent, bioaccumulative and toxic. Some restrictions on Lake Ontario sport fish consumption are due to accumulation of these compounds in sport fish. A screening assessment is insufficient to make conclusions about the aquatic environmental impacts of dioxins and furans from incinerators. A more detailed assessment, including sediment levels and biota residues, is required to better estimate actual risks. The size of the air emission also needs to be compared to other Ontario sources.

Sediments. Of a total of 57 chemicals, there were criteria for only 19 (Table C-3.2). Of these 19, most estimated concentrations were either background or close to the lowest effect level (LEL). The manganese concentration was between the LEL and severe effect level (SEL) but not significantly different than background and not likely of concern since manganese toxicity is similar to iron, a relatively low-hazard substance. Mercury was also between LEL and SEL and is evaluated below. Modelled chlorinated dioxin/furan toxicity equivalents barely exceeded the proposed Canadian Water Quality Sediment Guidelines (CCME 1994), by a factor of 1.7.

For chemicals without PSQGs or other criteria, no further evaluation was conducted. Further evaluation is possible for many organic chemicals however by calculating "no effect" concentrations from PWQOs or surrogate water quality criteria.

C-3.4.2 Risk Characterization: Landfills

Surface water. Aquatic criteria or surrogates were available or were developed for all air chemicals identified by IWA as having the potential to arise from landfilling (Table C-3.3).

Thirteen of these were derived from ASTER profiles (USEPA) using QSAR procedures. Criteria were derived by dividing the LOEC by 20 in the case of metals or 200 for organic chemicals (2000 in the case of bioaccumulative chemicals, i.e. those with log $K_{ow} > 4.0$).

The estimated surface water concentrations from direct deposition exceeded the water quality guidelines of four hydrocarbons: 3-methylheptane (risk ratio = 49), nonane (41), octane (93) and toluene (3). For the three alkanes, the estimated surface water concentrations were all < 2 ug/L. The mode of toxic action of alkanes is a non-specific, reversible narcotic action that generally occurs in the high ppb to low ppm range, depending on the water solubility of the compound. These estimated surface water levels thus are well below non-specific effect levels. In addition, fugacity fate modelling (ASTER, Mackay level I) indicated that these chemicals would, at equilibrium, partition 97 to 100% to air with the remainder partitioning to soil, sediment and/or water i.e. no predicted significant accumulation in biota. The higher alkanes did have low solubility and therefore a predicted high bioaccumulation factor, but bioaccumulation is unlikely given the high volatility and degradability.

Surface water toluene concentration, estimated at 2 ug/L, exceeded the PWQO (0.8 ug/L) by a factor of three. The estimated surface water concentration is ten times lower than the most sensitive species response found in the literature: lethality at 20 ppb in a 27-day early life stage fish exposure to constant concentrations of toluene. Again, a level III fugacity model (Mackay et al 1992a) indicated that toluene is volatile, degradable and generally not persistent. Also, 99% would remain in air, so ambient water concentrations would be much lower than estimated. The conclusion is that there is little potential for water quality impairment.

Leachate. Aquatic criteria or surrogates were available for all 57 chemicals identified by IWA as having the potential to arise from landfilling (Table C-3.3). Six of these were derived from QSAR toxicity data from ASTER profiles (USEPA), by dividing the LOEC by 20 for metals or 200 for organic compounds (2000 in the case of bioaccumulative chemicals - those having log $K_{ow} > 4.0$). Untreated, undiluted leachate exceeded criteria or surrogate criteria for 27 of 57 chemicals.

To provide a more realistic (less conservative) assessment, a second level evaluation was done. EEC were recalculated (Table C-3.4) using the leachate treatment system designed for the proposed EE11 landfill (IWA 1994e). The on-site sewage treatment system, including biological treatment was designed to reduce leachate concentrations to those acceptable in sewage as defined by the MISA model sewer use bylaw (OME 1988). Estimated removal was 70% for VOCs. Even higher values were found in the literature (SRC 1989). Where data existed, removal ranged from 75% to 99.99%, with most values >90%. Where data did not exist, a conservative default value of 50% removal was used.

The treated leachate would then be transported to a municipal sewage treatment plant. Design dilution in the plant was 233,500/230 m^3/d or over 1000 fold (IWA 1994e p A.6-42). Dilution in the receiving water of 10-fold was subsequently applied. After this second screening, all chemicals were at least two orders of magnitude lower than PWQOs.

Groundwater. All expected groundwater concentrations were less than surface water quality criteria (Table C-3.3). The highest ratios of EEC/EQC occurred for isovaleric acid, nitrates and phenols which reached about 0.1. Most of the remaining chemicals were 2 orders of magnitude less than their respective criteria.

C-3.4.3 Risk Characterization: Summary

At the screening level, neither incineration nor landfilling were predicted to have any demonstrable effects on local water quality in this scenario. While landfilling or incineration emit different amounts and types of chemicals, predicted water quality nearly always met respective criteria, and usually by a wide margin. Sediment quality criteria were met with three exceptions, although there were a large number of chemicals not evaluated due to a lack of criteria. Certainly no acute toxicity to aquatic life would be expected. The likelihood of chronic effects is also remote given the projected compliance with environmental quality criteria. Still, much of this scenario is based on efficient operation of well-designed disposal facilities, without upset.

Of nearly 90 chemicals potentially present in landfill emissions, those associated with air or groundwater were deemed to pose negligable risks to surface water. The three alkanes and toluene were considered too volatile and degradable to cause any impact. While many of the chemicals emitted from landfill vents are volatile and thus pose little risk to aquatic life the same is not true of untreated landfill leachates. It is clear that landfills can have influences on surrounding water if inadequately designed and operated. Landfill leachates can be highly enriched with organic chemicals arising from, or resistant to, microbial activity in the landfill. Release of inadequately treated leachate to surface waters has caused elevated chemical concentrations in surface waters draining some Ontario landfills. Leachate chemicals with high affinity for organic carbon were screened out by the IWA report and were not considered here (with the exception of lindane and pentachlorophenol in leachate which were assessed in this report). Many of these chemicals are also highly bioaccumulative, including organochlorine pesticides. This screening procedure is based on the premise that most of these chemicals will not escape from the landfill due to tight binding to organic material, or if in leachate, will be returned to landfills, agricultural soils, or combusted when sewage sludges are incinerated. Any chemicals with high K_{oc} which did escape to surface water would utimately accumulate in sediment. However there might be some concern due to bioaccumulation potential.

Incinerators discharge a surprising variety of organic chemicals which one might expect to burn completely. These would be widely dispersed. Only two groups of substances had some potential to impair water quality: mercury, and dioxins/furans. However, there is a general concern regarding accumulation of mercury and dioxins/furans in biota from diffuse sources. Further study is needed to evaluate the potential for deleterious effects from these substances.

Regarding effects on sediment quality due to incineration, two chemicals were of potential concern after twenty years of accumulation. Mercury was between the Lowest Effect Level and the Severe Effect Level sediment criteria. These concentrations would not, however, trigger a clean-up without a site-specific evaluation. Chlorinated dioxins were about two fold higher than the proposed Canadian Sediment Quality Guidelines (CCME 1994) but little different from the background soil quality and likewise would probably not require cleanup. Furthermore, evaluations making use of mobility and degradation characteristics, indicated that there is little likelihood of mercury and dioxins/furans reaching surface water in excess of their respective water quality criteria. Dioxins/furans, for example, are >90% retained in soils (Mackay et al 1992b). In addition it must be remembered that we have assumed worst case concentrations with no dilution or degradation. If there were still a concern regarding these chemicals, bioassessment of the site would be recommended, with risk management decisions based on the findings.

There is also the issue of mercury or dioxin/furan dispersal from incineration. The sediment/water to fish pathway for mercury was not specifically evaluated in the present risk assessment. Given that, in Ontario, fish consumption advisories (OMEE, OMNR 1990) are due largely to mercury contamination and occasionally dioxins/furans, some consideration should be given to province-wide loadings for these contaminants in any risk management decision.

Finally, there were difficulties with comparing landfilling and incineration in the current scenario. Emperical data were used for the incinerator whereas for landfilling, composite data from other locations were used. Incineration practice does benefit from widespread dispersal of contaminants, with only a small percentage of the emissions being deposited in the vicinity of the stack. Other issues of habitat destruction and the production of greenhouse gases should form an integral part of any risk management decisions. These aspects are discussed more fully in the terrestrial report.

C-3.5 Overall Uncertainty

There are many uncertainties in this evaluation. All of the assumptions used in this comparative risk scenario are discussed below along with a subjective estimate of the confidence in each component.

Exposure Assessment

Chemical identity.
See section of the report on air modelling.
Landfilling data as reported by IWA were extensive. However, the data were pre-screened based on human health and other criteria and it is possible that chemicals of aquatic concern were screened out. For example, there were no pesticides on the leachate list even though these are found in landfills (FMI 1994). Screening procedures used here were mobility and frequency of detection. This reduced the list from 169 to 58 (IWA 1994d p QD-80). Generally, uncertainty is low.

Dispersion modelling
See section of the report on air modelling.
Landfill modelling of volatile organic compounds is generally done at 25°C with the result that chemicals would be less volatile in winter. The deposition velocity constant of 0.0065 mls developed by McKone and Daniels (1991) may not be ideal for calculating deposition of VOCs to soil. Although it does account for wet scavenging, it was developed specifically for absorption to vegetation. Generally, uncertainty is low for most chemicals but moderate for mercury.

Surface water concentrations
Estimation of surface water concentrations used averaging procedures. This was based largely on the assumption that chemical concentrations would likely not reach acutely toxic levels and that time lags in the system, such as soil runoff, would produce an averaging effect. Even though many of these assumptions were conservative, it is possible that more extreme concentrations could occur. For example, maximum annual average GLC was used but daily averages were higher. The effects of snow and cold temperature are not considered. One of the findings of acid rain research is that spring snow melt releases the winter's accumulated acid load into streams. Similar pulse-loading of accumulated chemicals certainly occurs as well.

The assumption that vent chemicals were all volatile, such that the runoff and sediment concentrations would not be affected, may be overly simplistic.

Disposal of incinerator ash has not been addressed. It was assumed that bottom ash from incinerators would be landfilled and this was accounted for in the landfill evaluation. Fly ash, currently treated as hazardous waste, would be disposed of in a secure hazardous landfill.

Hazard assessment

The use of Canadian environmental quality criteria is more conservative than other hazard or effects estimates such as USEPA ambient water quality criteria, "no effect concentrations" or EC20s for population endpoints such as growth and reproduction as proposed by Suter 1993. Surrogate criteria for organic chemicals developed for this assessment, and some PWQOs have large safety/uncertainty factors. Hazard assessments thus ranged from very good for well-studied contaminants to uncertain for where surrogate criteria were derived. There is something of a paradox here since chemicals without criteria may well be of minor concern since they have not found their way onto priority chemical lists even after 25 years experience. Still, the surrogate criteria which were derived had very conservative uncertainty factors applied to them. While it is recommended that criteria be developed for some chemicals, generally the hazard assessment component was not the area of difficulty.

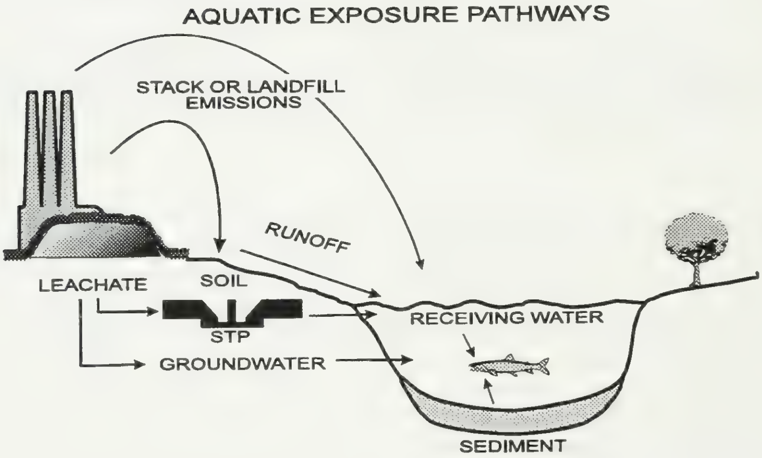
Risk/Hazard characterization

This risk/hazard characterization assumed that background concentrations are unimportant. This would not be true for some sites. The Durham EE11 site reported water quality surveys (IWA 1994c) which indicated that while PWQO were generally met at two monitoring stations, some exceedences of water quality criteria did occasionally occur for pH, ammonia, coliforms, phosphorous, iron, silver, cadmium, copper, vanadium, cyanide and phenols (IWA 1994c p D-3.8). While an exceedence of water quality criteria does not necessarily indicate detrimental effects, it does indicate that the assimilative capacity of the receiving water needs to be considered in a site-specific assessment.

Finally, this evaluation has only examined the impact of a single source on the local area. The consequences province-wide, of a widespread switch to incineration or landfilling were not evaluated in the current comparative risk assessment. This would be of considerable concern if this report were used in policy development.

The overall level of uncertainty is moderate due to the sum of the uncertainties above, but it errs on the side of protection of the environment.

Figure C-2.1. Conceptual model of the landfill/incinerator site.



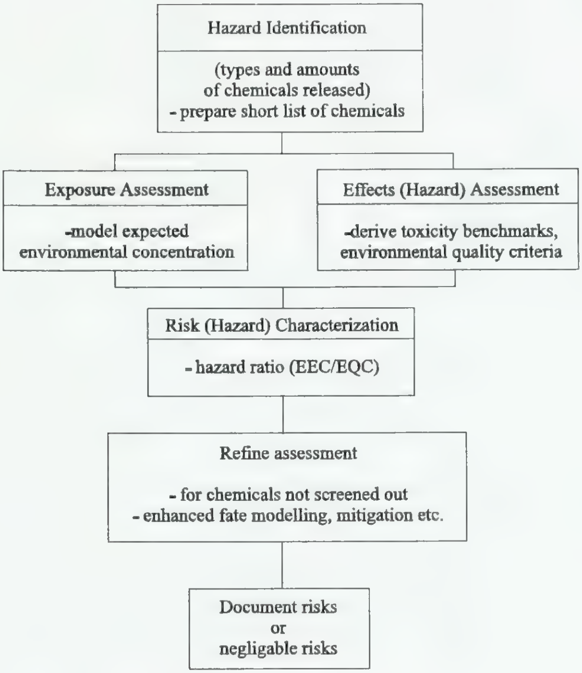

Hazard Identification
(types and amounts of chemicals released)
- prepare short list of chemicals
Exposure Assessment
-model expected environmental concentration
Effects (Hazard) Assessment
-derive toxicity benchmarks, environmental quality criteria
Risk (Hazard) Characterization
- hazard ratio (EEC/EQC)
Refine assessment
- for chemicals not screened out
- enhanced fate modelling, mitigation etc.
Document risks
or
negligable risks

Figure C-3.1. Screening Assessment Scheme.

Table C-3.1 Predicted surface water chemical concentrations arising from incineration.

Predicted surface water concentrations arising from incineration

	Stack Testing Parameters (1994)			annual avg GLC factor 7.53e-02 (ug/m3)	deposit to watershed		direct deposit to water, plus runoff		Water conc (ug/L)	Environ. Quality Criteria (ug/L)	ratio to EQC
	Uncorrected Emission rate (ug/s)	Corrected Emission rate (ug/s)	Corrected Emission rate (g/s)		dry deposition (ug/m2.d)	wet deposition (ug/m2.d)	water area assume 4%	runoff assume 40%			
Particulate emissions	81000.00	117739.88	1.18E-01	8.87E-03	1.53E+01	3.63E+00	7.58E-01	7.28E+00	2.01E+01	15000	0.00
Sulphur Dioxide	723142.00	1051143.83	1.05E+00	7.92E-02	1.37E+02	3.24E+01	6.77E+00	6.50E+01	1.79E+02		
Nitrogen Oxides	26203249.00	19044242.00	1.90E+01	1.43E+00	2.48E+03	5.87E+02	1.23E+02	1.18E+03	3.25E+03		
Hydrogen Chloride	544024.00	790781.72	7.91E-01	5.95E-02	1.03E+02	2.44E+01	5.09E+00	4.89E+01	1.35E+02		
Aluminum	752.00	1093.09	1.09E-03	8.23E-05	1.42E-01	3.37E-02	7.04E-03	6.76E-02	1.86E-01	75	0.00
Antimony	29.00	42.15	4.22E-05	3.17E-06	5.48E-03	1.30E-03	2.71E-04	2.61E-03	7.19E-03	20	0.00
Arsenic	15.00	21.80	2.18E-05	1.64E-06	2.84E-03	6.72E-04	1.40E-04	1.35E-03	3.72E-03	5	0.00
Barium	40.00	58.14	5.81E-05	4.38E-06	7.57E-03	1.79E-03	3.74E-04	3.59E-03	9.92E-03	0.25	0.04
Beryllium	6.30	9.16	9.16E-06	6.90E-07	1.19E-03	2.82E-04	5.90E-05	5.66E-04	1.56E-03	1100	0.00
Boron	2026.00	2944.95	2.94E-03	2.22E-04	3.83E-01	9.08E-02	1.90E-02	1.82E-01	5.02E-01	200	0.00
Cadmium	49.00	71.23	7.12E-05	5.36E-06	9.27E-03	2.20E-03	4.59E-04	4.40E-03	1.22E-02	0.5	0.02
Calcium	36389.00	52894.28	5.29E-02	3.98E-03	6.88E+00	1.63E+00	3.41E-01	3.27E+00	9.02E+00		
Chromium	102.20	148.56	1.49E-04	1.12E-05	1.93E-02	4.58E-03	9.56E-04	9.18E-03	2.53E-02	100	0.00
Cobalt	109.60	159.31	1.59E-04	1.20E-05	2.07E-02	4.91E-03	1.03E-03	9.85E-03	2.72E-02	0.6	0.05
Copper	165.70	240.86	2.41E-04	1.81E-05	3.13E-02	7.42E-03	1.55E-03	1.49E-02	4.11E-02	5	0.01
Iron	4431.00	6440.81	6.44E-03	4.85E-04	8.38E-01	1.99E-01	4.15E-02	3.98E-01	1.10E+00	300	0.00
Lead	467.00	678.82	6.79E-04	5.11E-05	8.83E-02	2.09E-02	4.37E-03	4.20E-02	1.16E-01	5	0.02
Lithium	87.80	127.62	1.28E-04	9.61E-06	1.66E-02	3.93E-03	8.22E-04	7.89E-03	2.18E-02	15	0.00

Magnesium	688.00	1000.06	1.00E-03	7.53E-05	1.30E-01	3.08E-02	6.44E-03	6.18E-02	1.71E-01		
Manganese	202.00	293.62	2.94E-04	2.21E-05	3.82E-02	9.05E-03	1.89E-03	1.81E-02	5.01E-02	14	0.00
Mercury	**8690.00**	**12631.60**	**1.26E-02**	9.51E-04	1.64E+00	3.89E-01	8.13E-02	7.81E-01	2.15E+00	**0.2**	**10.77**
Molybdenum	80.60	117.16	1.17E-04	8.82E-06	1.52E-02	3.61E-03	7.54E-04	7.24E-03	2.00E-02	10	0.00
Nickel	1648.00	2395.50	2.40E-03	1.80E-04	3.12E-01	7.38E-02	1.54E-02	1.48E-01	4.09E-01	25	0.02
Phosphorus	871.00	1266.07	1.27E-03	9.53E-05	1.65E-01	3.90E-02	8.15E-03	7.82E-02	2.16E-01	10	0.02
Selenium	36.00	52.33	5.23E-05	3.94E-06	6.81E-03	1.61E-03	3.37E-04	3.23E-03	8.93E-03	100	0.00
Silicon	816782.00	1187256.94	1.19E+00	8.94E-02	1.54E+02	3.66E+01	7.64E+00	7.34E+01	2.03E+02		
Silver	63.40	92.16	9.22E-05	6.94E-06	1.20E-02	2.84E-03	5.93E-04	5.70E-03	1.57E-02	0.1	0.16
Sodium	14579.00	21191.72	2.12E-02	1.60E-03	2.76E+00	6.53E-01	1.36E-01	1.31E+00	3.62E+00	20000	0.00
Strontium	21.00	30.53	3.05E-05	2.30E-06	3.97E-03	9.41E-04	1.97E-04	1.89E-03	5.21E-03	0.007	0.74
Tin	332.90	483.90	4.84E-04	3.64E-05	6.30E-02	1.49E-02	3.12E-03	2.99E-02	8.26E-02	250	0.00
Titanium	72.00	104.66	1.05E-04	7.88E-06	1.36E-02	3.23E-03	6.74E-04	6.47E-03	1.79E-02	102	0.00
Vanadium	22.00	31.98	3.20E-05	2.41E-06	4.16E-03	9.86E-04	2.06E-04	1.98E-03	5.46E-03	7	0.00
Zinc	3775.00	5487.26	5.49E-03	4.13E-04	7.14E-01	1.69E-01	3.53E-02	3.39E-01	9.36E-01	20	0.05
Fluoride	14.60	21.22	2.12E-05	1.60E-06	2.76E-03	6.54E-04	1.37E-04	1.31E-03	3.62E-03	200	0.00
1,2-Dichlorobenzene	1.41	2.05	2.05E-06	1.54E-07	2.67E-04	6.32E-05	1.32E-05	1.27E-04	3.50E-04	2.5	0.00
1,3-Dichlorobenzene	1.33	1.93	1.93E-06	1.46E-07	2.52E-04	5.96E-05	1.24E-05	1.19E-04	3.30E-04	2.5	0.00
1,4-Dichlorobenzene	16.30	23.69	2.37E-05	1.78E-06	3.08E-03	7.30E-04	1.53E-04	1.46E-03	4.04E-03	4	0.00
1,2,3-Trichlorobenzene	1.07	1.56	1.56E-06	1.17E-07	2.02E-04	4.79E-05	1.00E-05	9.61E-05	2.65E-04	0.9	0.00
1,2,4-Trichlorobenzene	2.67	3.88	3.88E-06	2.92E-07	5.05E-04	1.20E-04	2.50E-05	2.40E-04	6.62E-04	0.5	0.00
1,2,3,4-Tetrachlorobenzene	1.15	1.67	1.67E-06	1.26E-07	2.18E-04	5.15E-05	1.08E-05	1.03E-04	2.85E-04	0.65	0.00
1,2,3,5- & 1,2,4,5-Tetrachlorobenzene	2.70	3.92	3.92E-06	2.96E-07	5.11E-04	1.21E-04	2.53E-05	2.43E-04	6.70E-04	0.1	0.01

Pentachlorobenzene	2.56	3.72	3.72E-06	2.80E-07	4.84E-04	1.15E-04	2.40E-05	2.30E-04	6.35E-04	0.03	0.02
Hexachlorobenzene	0.28	0.41	4.07E-07	3.06E-08	5.30E-05	1.25E-05	2.62E-06	2.52E-05	6.94E-05	0.0065	0.01
2,5-2,4-Dichlorophenol	3.85	5.60	5.60E-06	4.21E-07	7.28E-04	1.72E-04	3.60E-05	3.46E-04	9.55E-04	0.2	0.00
2,6-Dichlorophenol	1.15	1.67	1.67E-06	1.26E-07	2.18E-04	5.15E-05	1.08E-05	1.03E-04	2.85E-04	0.2	0.00
2,3,4-Trichlorophenol	2.69	3.91	3.91E-06	2.94E-07	5.09E-04	1.21E-04	2.52E-05	2.42E-04	6.67E-04	18	0.00
2,4,5-Trichlorophenol	1.26	1.83	1.83E-06	1.38E-07	2.38E-04	5.65E-05	1.18E-05	1.13E-04	3.12E-04	18	0.00
2,4,6-Trichlorophenol	5.30	7.70	7.70E-06	5.80E-07	1.00E-03	2.37E-04	4.96E-05	4.76E-04	1.31E-03	18	0.00
2,3,4,6-Tetrachlorophenol	3.29	4.78	4.78E-06	3.60E-07	6.22E-04	1.47E-04	3.08E-05	2.96E-04	8.16E-04	1	0.00
Pentachlorophenol	1.90	2.76	2.76E-06	2.08E-07	3.59E-04	8.51E-05	1.78E-05	1.71E-04	4.71E-04	0.5	0.00
Tetralin	9.43	13.71	1.37E-05	1.03E-06	1.78E-03	4.22E-04	8.82E-05	8.47E-04	2.34E-03		
Biphenyl	56.43	82.03	8.20E-05	6.18E-06	1.07E-02	2.53E-03	5.28E-04	5.07E-03	1.40E-02	0.2	0.07
o-Terphenyl	1.55	2.25	2.25E-06	1.70E-07	2.93E-04	6.94E-05	1.45E-05	1.39E-04	3.84E-04		
2-Methylnaphthalene	5.20	7.56	7.56E-06	5.69E-07	9.84E-04	2.33E-04	4.87E-05	4.67E-04	1.29E-03	2	0.00
1-Methylnaphthalene	1.86	2.70	2.70E-06	2.04E-07	3.52E-04	8.33E-05	1.74E-05	1.67E-04	4.61E-04	2	0.00
2-Methylfluorene	289.00	420.08	4.20E-04	3.16E-05	5.47E-02	1.29E-02	2.70E-03	2.60E-02	7.17E-02	0.07	1.07
1-Methylpyrene	0.32	0.47	4.65E-07	3.50E-08	6.05E-05	1.43E-05	2.99E-06	2.87E-05	7.94E-05	0.01	0.01
Naphthalene	26.52	38.55	3.85E-05	2.90E-06	5.02E-03	1.19E-03	2.48E-04	2.38E-03	6.58E-03	7	0.00
Acenaphthylene	0.53	0.77	7.70E-07	5.80E-08	1.00E-04	2.37E-05	4.96E-06	4.76E-05	1.31E-04	0.01	0.01
Acenaphthene	1.80	2.62	2.62E-06	1.97E-07	3.40E-04	8.06E-05	1.68E-05	1.62E-04	4.46E-04	*6*	0.00
Fluorene	0.90	1.31	1.31E-06	9.85E-08	1.70E-04	4.03E-05	8.42E-06	8.08E-05	2.23E-04	0.2	0.00
Phenanthrene	4.29	6.24	6.24E-06	4.70E-07	8.11E-04	1.92E-04	4.01E-05	3.85E-04	1.06E-03	0.03	0.04
Fluoranthene	0.76	1.10	1.10E-06	8.32E-08	1.44E-04	3.40E-05	7.11E-06	6.83E-05	1.88E-04	0.0008	0.24
Pyrene	0.71	1.03	1.03E-06	7.77E-08	1.34E-04	3.18E-05	6.64E-06	6.38E-05	1.76E-04	*0.025*	0.01
Benzo(b)fluoranthene	0.10	0.15	1.45E-07	1.09E-08	1.89E-05	4.48E-06	9.36E-07	8.98E-06	2.48E-05	0.01	0.00

Benzo(a)fluorene	0.52	0.76	7.56E-07	5.69E-08	9.84E-05	2.33E-05	4.87E-06	4.67E-05	1.29E-04	0.01	0.01
Benzo(b)fluorene	0.36	0.52	5.23E-07	3.94E-08	6.81E-05	1.61E-05	3.37E-06	3.23E-05	8.93E-05	0.01	0.01
Dioxins & furans - total TEQ	**0.01**	**0.01**	**8.14E-09**	6.13E-10	1.06E-06	2.51E-07	5.24E-08	5.03E-07	1.39E-06	***3.80E-07***	**3.66**
Total Chlorobiphenyl (PCBs)	1.29	1.88	1.88E-06	1.41E-07	2.44E-04	5.78E-05	1.21E-05	1.16E-04	3.20E-04	0.001	0.32
Chloromethane	991.00	1440.50	1.44E-03	1.08E-04	1.87E-01	4.44E-02	9.27E-03	8.90E-02	2.46E-01	700	0.00
Bromomethane	227.00	329.96	3.30E-04	2.48E-05	4.29E-02	1.02E-02	2.12E-03	2.04E-02	5.63E-02	0.9	0.06
Dibromoethane	2.70	3.92	3.92E-06	2.96E-07	5.11E-04	1.21E-04	2.53E-05	2.43E-04	6.70E-04	75	0.00
Dichlorodifluoromethane	489.00	710.80	7.11E-04	5.35E-05	9.25E-02	2.19E-02	4.58E-03	4.39E-02	1.21E-01	68	0.00
Trichlorofluoromethane	1086.00	1578.59	1.58E-03	1.19E-04	2.05E-01	4.87E-02	1.02E-02	9.76E-02	2.69E-01	32	0.01
Vinyl Chloride	275.00	399.73	4.00E-04	3.01E-05	5.20E-02	1.23E-02	2.57E-03	2.47E-02	6.82E-02	400	0.00
1,1-Dichloroethene	3.60	5.23	5.23E-06	3.94E-07	6.81E-04	1.61E-04	3.37E-05	3.23E-04	8.93E-04	40	0.00
Trichloroethene	9.30	4.50	4.50E-06	3.39E-07	5.86E-04	1.39E-04	2.90E-05	2.78E-04	7.68E-04	20	0.00
I,I,I-Trichloroethane	9.00	13.08	1.31E-05	9.85E-07	1.70E-03	4.03E-04	8.42E-05	8.08E-04	2.23E-03	10	0.00
Tetrachloroethene	3.00	20.70	2.07E-05	1.56E-06	2.69E-03	6.38E-04	1.33E-04	1.28E-03	3.53E-03	50	0.00
Benzene	17123.00	24889.63	2.49E-02	1.87E-03	3.24E+00	7.67E-01	1.60E-01	1.54E+00	4.25E+00	100	0.04
Toluene	319.00	463.69	4.64E-04	3.49E-05	6.03E-02	1.43E-02	2.99E-03	2.87E-02	7.91E-02	0.8	0.10
Ethylbenzene	6.50	9.45	9.45E-06	7.11E-07	1.23E-03	2.91E-04	6.08E-05	5.84E-04	1.61E-03	8	0.00
Total Xylenes	3810.00	5538.13	5.54E-03	4.17E-04	7.21E-01	1.71E-01	3.57E-02	3.42E-01	9.45E-01	2	0.47

Table of constants	Name			
emission rate to GLC ann avg	er2glc	7.53E-02		max annual average
deposition velocity	Vd	0.02	m/s	
seconds to days conversion	s2d	86400		
low flow adjusted for land area	Q_adj	4.00E-01	L/m2.d	was 400 m3/km2.d
wet deposition	wetdep	3.08E+01	ug/m2.d per g/s emission (recalc from 180 kg/a over 16 km2)	

Notes

1. wet scavenging is based on R. Bloxam's estimate of 180 kg/a per g/s of emissions
2. direct wet deposition is calc as the above /365 * water/land plus conversion factors
3. BC criteria in italics. Values in courier font indicate use of the lowest BC EQC (for anthracene)
the methylfluorene criteria is the BC fluorene criteria divided by 3 for methyl grp
4. All values corrected for stack temp to ground level (25 Celsius)
5. Peel MSW Incinerator - Maxima from 1993 & 1994 tests + positives from Hartford for some Peel ND
6. uses all factors adjusted to m2

Table C-3.2 Additional screening of contaminants with hazard ratio greater than one.

Additional screening of contaminants with EEC/EQC >1

	Stack Testing Parameters (1994)			annual avg GLC factor 1.12E-02 (ug/m3)	**deposit to watershed**		**direct deposit to water, plus runoff**		Water conc (ug/L)	Environ. Quality Criteria (ug/L)	ratio to EQC
	Uncorrected Emission rate (ug/s)	Corrected Emission rate (ug/s)	Corrected Emission rate (g/s) comply A-7		dry deposition (ug/m2.d)	wet deposition (ug/m2.d)	water area assume 4%	runoff assume 40%			
Mercury	8690	12632	**1.71E-03**	**1.91E-05**	3.30E-02	5.26E-02	3.42E-03	**1.23E-02**	**1.97E-02**	0.2	0.10
Dioxins & furans - total TEQ	0.005602	0.008	**4.23E-09**	**4.74E-11**	8.19E-08	1.31E-07	8.50E-09	8.16E-08	2.25E-07	*3.80E-07*	0.59
2-Methylfluorene	289	420	4.20E-04	**4.70E-06**	8.13E-03	1.29E-02	8.43E-04	8.09E-03	2.23E-02	0.07	0.33

Table of constants	**Name**		
emission rate to GLC ann avg	**er2glc**	**7.53E-02**	**max annual average**
		3.19E-05	**min annual average**
		1.12E-02	**uniform grid avg**
deposition velocity	**Vd**	**0.02**	**m/s**
seconds to days conversion	**s2d**	**86400**	
low flow adjusted for land area	**Q_adj**	**4.00E-01**	**L/m2.d was 400 m3/km2.d**
wet deposition	**wetdep**	**3.08E+01**	**ug/m2.d per g/s emission (recalc from 180 kg/a over 16 km2)**

Notes

Numbers in bold changed due to the following:

- mercury and dioxins/furans emissions were reduced by factors of 0.135 and 0.52 resp, to comply with A-7 incineration guidelines
- GLC were dispersed over the area of the entire watershed (uniform grid average rather than annual maximum average)
- mercury runoff was reduced to 15% (G. Mierle, pers. com.)
- waterborne mercury concentration reduced by 0.5 to equate to methylmercury

Table C-3.3 Predicted sediment chemical concentrations arising from incineration.

Incinerator emissions PEEL DATA	POIAvg (ug/m3)	drydeprate (ug/m2/yr)	5cmadd (ug/g)	Maxsc5cm (ug/g)	OTR98r	POI (ug/m3)	drydep (ug/m2.a)	wetdep	5cmadd (ug/g)	soilmax	% increase to		PSQG			soilmax>		
Chemical											OTR 50	OTR 98	NEL	LEL	SEL	NEL	LEL	SEL
													ug/g dry weight					
Al	0.00384	242.2	0.1	16000.1	3.00E+04	9.63E-03	6.07E+02	0.49	2.43E-01	3.00E+04								
Sb	0.000131	8.26	0	0.15	4.30E-01	3.71E-04	2.34E+01	0.02	9.37E-03	4.39E-01	0.06	0.02						
As	0.0000846	5.34	0	4	1.10E+01	1.92E-04	1.21E+01	0.01	4.85E-03	1.10E+01	0.00	0.00	4.2	6	33	hit	hit	-
Ba	0.000191	12.05	0	70	1.60E+02	5.12E-04	3.23E+01	0.03	1.29E-02	1.60E+02	0.00	0.00						
Be	0.0000263	1.66	0	0.45	1.10E+00	5.89E-05	3.71E+00	0	1.49E-03	1.10E+00	0.00	0.00						
B	0.01355	854.63	0.34	14.34	3.00E+01	2.81E-02	1.78E+03	1.33	7.11E-01	3.07E+01	0.05	0.02						
Cd	0.000226	14.25	0.01	0.26	7.10E-01	6.27E-04	3.96E+01	0.03	1.58E-02	7.26E-01	0.06	0.02	1.1	0.6	10	-	hit	-
Ca	0.166	10469.95	4.19	10004.19	5.50E+04	4.65E-01	2.94E+04	23.8	1.18E+01	5.50E+04	0.00	0.00						
Cr	0.0005315	33.52	0.01	25.01	5.80E+01	9.52E-03	6.00E+02	0.07	2.40E-01	5.82E+01	0.01	0.00	31	26	110	hit	hit	-
Co	0.000419	26.43	0.01	7.51	1.60E+01	1.40E-03	8.84E+01	0.07	3.54E-02	1.60E+01	0.00	0.00		50			-	
Cu	0.000368	23.21	0.01	16.01	4.10E+01	9.91E-04	6.25E+01	0.11	2.50E-02	4.10E+01	0.00	0.00	25	16	110	hit	hit	-
Fe	0.0216	1362.36	0.54	20000.54	3.50E+04	5.67E-02	3.58E+03	2.9	1.43E+00	3.50E+04	0.00	0.00	31200	20000	40000	hit	hit	-
Pb	0.0023	145.07	0.06	22.06	4.50E+01	3.16E-02	2.00E+03	0.31	7.98E-01	4.58E+01	0.04	0.02	23	31	250	hit	hit	
Li	0.0003315	20.91	0.01	0.01		7.42E-04	4.68E+01	0.06	1.88E-02									
Mg	0.00239	150.74	0.06	6000.06	2.00E+04	6.50E-03	4.10E+02	0.45	1.64E-01	2.00E+04	0.00	0.00						
Mn	0.001123	70.83	0.03	900.03	1.30E+03	2.59E-03	1.63E+02	0.13	6.53E-02	1.30E+03	0.00	0.00	400	460	1100	hit	hit	hit
Hg	0.0212	1337.13	0.53	0.59	1.30E-01	4.86E-02	3.07E+03	5.68	1.23E+00	1.36E+00	20.49	9.46	0.1	0.2	2	hit	hit	
Ni	0.0057495	362.63	0.15	13.15	3.80E+01	2.11E-02	1.33E+03	1.08	5.33E-01	3.85E+01	0.04	0.01	31	16	75	hit	hit	-
P	0	0	0	540	9.00E+02	2.60E-03	1.64E+02	0.57	6.58E-02	9.00E+02	0.00	0.00		600	2000	hit	hit	-
Se	0.00015725	9.92	0	0.4	9.30E-01	4.61E-04	2.91E+01	0.02	1.16E-02	9.42E-01	0.03	0.01						
Si	3.5	220752	88.3	88.3		7.87E+00	4.96E+05	534.27	1.99E+02									
Ag	0.000263	16.59	0.01	0.13	2.70E-01	5.89E-04	3.71E+01	0.04	1.49E-02	2.85E-01	0.12	0.06						
Na	0.07415	4676.79	1.87	226.87	6.60E+02	1.87E-01	1.18E+04	9.54	4.71E+00	6.65E+02	0.02	0.01						
Sr	0.00009545	6.02	0	28	6.40E+01	2.69E-04	1.70E+01	0.01	6.79E-03	6.40E+01	0.00	0.00						

Sn	0.000629	39.67	0.02	0 02		1.42E-03	8.96E+01	0.22	3.59E 02	3.59E-02								
Ti	0 00036	22 71	0.01	3200.01	5.20E+03	1.05E-03	6.62E+01	0.05	2.65E-02	5.20E+03	0.00	0 00						
V	0 0001025	6 46	0	40	7.70E+01	2.30E-04	1.45E+01	0.01	5.82E-03	7.70E+01	0 00	0 00						
Zn	0 01465	924	0 37	60.37	1.20E+02	4 83E-02	3.05E+03	2.47	1 22E+00	1 21E+02	0 02	0 01	65	120	820	hit	hit	
F	0.0000686	4.33	0	18	6.10E+01	1.54E-04	9.69E+00	0.01	3.88E-03	6.10E+01	0.00	0.00						
1,2-Dichlorobenzene	0	0	0	0	3.00E-06	2.48E-04	5.08E-01	0	2.03E-04	2.06E-04		67.69						
1,3-Dichlorobenzene	0	0	0	0	3.00E-06	7.87E-05	1.61E-01	0	6.45E-05	6.75E-05		21.52						
1,4-Dichlorobenzene	0	0	0	0	4.60E-05	1.59E-04	3 25E-01	0	1.30E-04	1.76E-04		2.83						
1,2,3-Trichlorobenzene	0	0	0	0			0.00	0	0.00	0.00								
1,2,4-Trichlorobenzene	0.00000893	0.56	0	0			0.00	0	0.00	0.00								
1,3,5 Trichlorobenzene	0	0	0	0			0.00	0	0.00	0.00								
1,2,3,4-Tetrachlorobenzene	0	0	0	0			0.00	0	0.00	0.00								
1,2,3,5 & 1,2,4,5-T4Clbenzene	0	0	0	0			0.00	0	0 00	0.00								
Pentachlorobenzene	0	0	0	0		1.48E-05	3.04E-02	0	1.22E-05	1.22E-05								
2,3,4-Trichlorophenol	0	0	0	0	6 00E 03		0.00	0	0.00	6.00E-03		0.00						
2,3,4,6-Tetrachlorophenol	0	0	0	0	1.40E-02		0.00	0	0.00	1.40E-02		0.00						
Pentachlorophenol	0 00000588	0 37	0	0	1 40E-02		0.00	0	0.00	1.40E-02		0.00						
Tetralin	0	0	0	0		8.96E 05	1.84E-01	0	7.35E-05	7.35E-05								
Quinoline	0	0	0	0		1.54E-05	3.15E-02	0	1.26E-05	1.26E-05								
2 Methylnaphthalene	0	0	0	0	3.00E 03	6 66E 05	1.36E-01	0	5.46E-05	3.05E-03		0.02						
1-Methylnaphthalene	0	0	0	0	6 00E-03	2.43E-05	4.99E-02	0	1.99E-05	6.02E-03		0 00						
Biphenyl	0	0	0	0		8.96E-05	1.84E-01	0	7.35E-05	7.35E-05								
Naphthalene	0 000138	8.7	0	0	6.00E-03	3.30E-04	6.77E-01	0	2.71E-04	6.27E-03		0.05						
Acenaphthylene	0	0	0	0 01	3.00E-02	2 50E-05	5.12E-02	0	2.05E-05	3.00E-02	0.00	0.00						
Acenaphthene	0	0	0	0	6.00E-03	6.40E-06	1.31E-02	0	5.25E-06	6.01E-03		0.00						
Fluorene	0	0	0	0	9.00E-03	1.41E-05	2.89E-02	0	1.15E-05	9.01E-03		0.00		0 19	160	hit	-	-
Phenenthrene	0	0	0	0	9.20E-02	7.42E-05	1.52E-01	0	6.09E-05	9.21E-02		0.00		0.56	950	hit	-	-

Anthracene	0	0	0	0	6.00E-03	9.60E-06	1.97E-02	0	7.87E-06	6.01E-03		0.00		0.22	370	hit	-	-
Fluoranthene	0	0	0	0	1.40E-01	8.45E-05	1.73E-01	0	6.93E 05	1.40E-01		0.00		0.75	1020	hit	-	-
Pyrene	0	0	0	0	1.10E-01	4.54E-05	9.31E-02	0	3.73E-05	1.10E-01		0.00		0.49	850	hit	-	-
Benzo(a)pyrene	0 00000233	0.15	0	0	3.90E 02		0.00	0	0.00	3.90E-02		0.00		0.37	1440	hit	-	
total chl. dioxins / furans	0 0003985	25.13	0.01	0 01	4.80E-06	4.85E-08	9 94E-05	0	3.98E-08	4.84E-06	0.02	0.01	*1.70-06*			hit		

OTR98 is detection limit or actual data, whereas the OTR50 is well below detection limit
CCME draft sediment guideline is 1.7E-06 ug/g
POI is point of impingement standard - similar to Ground Level Concenntration
maxsc is maximum soil concentration calculated by adding the accumulated contaminant load to the OTR98
50sc is soil concentration calculated by adding the accumulated contaminant load to the OTR50
wet deposition was pasted from peelmax3.xls and multiplied by 365 to give yearly depostion
PSQG (Provincial Sediment Quality Guidelines)
NEL - no effect level, LEL - lowest effect level, SEL - severe effect level
No effect concentrations for metals are based on the pre-colonial horizon; for organics the PWQO is multiplied by the organic carbon partition coefficient

Table C-3.4 Predicted surface water, leachate and groundwater chemical concentrations arising from landfilling.

Landfilling									
Inputs --------------->	CAS no	wet and dry deposition ug/L now from QD19djs3	runoff ug/L	leachate ug/L	reasonable worst case groundwater ug/L	PWQO ug/L	ratio to PWQO		
Chemical							direct dep	leachate	groundwater
acetic acid	64-19-7		0	4.30E+04	1.20E+00	395		**108.86**	0.00
acetone	67-64-1	7.60E-01		3.90E+00		50	0.02	0.08	
acrylonitrile	107-13-1	3.91E-03				13	0.00		
benzene	71-43-2	1.01E-01		8.62E+00	2.60E-02	100	0.00	0.09	0.00
bromochloromethane	74-97-5		0	1.37E+00	2.80E-04	335		0.00	0.00
bromodichloromethane	74-97-5	6.44E-01			3.10E-02	200	0.00		0.00
butan-2-ol	78-92-2	5.61E+00				475	0.01		
butane	106-97-8	1.88E+00				6.395	0.29		
butyl benzyl phthalate	85-68-7		0	1.51E+00		0.2		**7.55**	
butylacetate (used tert-)	540-88-5	7.97E-02				1635	0.00		
butyric acid	107-92-6		0	5.67E+04	9.30E+00	325		**174.46**	0.03
cadmium	7440-43-9	3.99E-07				0.1	0.00		
camphor	76-22-2		0	5.40E+00		125		0.04	
carbon disulphide	75-15-0	1.02E-01				475	0.00		
carbon tetrachloride	56-23-5	2.31E-03				3.4	0.00		
carbonyl sulphide	463-58-1	9.59E-01				2	0.48		
chlorobenzene	108-90-7	2.36E-03		5.10E-01		15	0.00	0.03	
chlorodifluoromethane	75-45-6	1.40E+00				310.54	0.00		

chloroethane	75-00-3	1.44E-01		4.40E-01	5.90E-03	105.35	0.00	0.00	0.00
chloroform	67-66-3	8.85E-03		1.04E+00	2.70E-02	10	0.00	0.10	0.00
chloromethane	74-87-3	9.13E-02		1.22E+00	7.00E-02	700	0.00	0.00	0.00
chromium	7440-47-3	1.39E-05				100	0.00		
chromium VI	7440-47-3		0	8.80E+01		2		**44.00**	
copper	7440-50-8	5.10E-05				5	0.00		
cresol, o-	95-48-7		0	3.38E+00	2.20E-04	1		**3.38**	0.00
cresol, p-	106-44-5		0	6.00E+02	5.30E-02	1		**600.00**	0.05
cyclohexane	110-82-7	4.49E-01				150	0.00		
decane	124-18-5	1.92E+00				90	0.02		
di-n-butyl phthalate	84-74-2		0	1.13E+00		4		0.28	
di-n-octylphthalate	117-84-0		0	2.21E+00		0.2		**11.05**	
dibromoethane, 1,2-	106-93-4	2.50E-03				75	0.00		
dichlorobenzene, 1,2-	95-50-1	1.48E-03		8.46E-01		2.5	0.00	0.34	
dichlorobenzene, 1,3-	541-73-1	5.33E-03		4.09E-01		2.5	0.00	0.16	
dichlorobenzene, 1,4-	106-46-7	1.26E-01		8.60E-01	6.80E-03	4	0.03	0.22	0.00
dichlorodifluoromethane	75-71-8	5.47E+00		6.24E+00		68.205	0.08	0.09	
dichloroethane, 1,1-	75-34-3	1.81E+00		5.28E+01	4.80E+00	200	0.01	0.26	0.02
dichloroethane, 1,2-	107-06-2	2.07E-02		4.70E-01	2.80E-02	100	0.00	0.00	0.00
dichloroethylene, 1,1-	75-35-4	1.39E-01		1.21E+00	4.20E-02	40	0.00	0.03	0.00
dichloroethylene, 1,2-	156-59-2	1.64E-01		5.03E-01	7.50E-01	200	0.00	0.00	0.00
dichlorofluoromethane	75-43-4	8.57E-01				152.95	0.01		
dichloropropane, 1,2-	78-87-5	3.66E-02				48.458	0.00		

diethyl ether	60-29-7		0	2.95E+00		825		0.00	
diethyl phthalate	84-66-2		0	2.51E+01		0.2		**125.50**	
dimethyl phthalate	131-11-3		0	1.31E+00		0.2		6.55	
dimethyl sulphide	75-18-3	3.79E-01		1.03E+01	2.90E-02	353.91	0.00	0.03	0.00
dimethyldisulphide	624-92-0	3.59E-03				0.2	0.02		
dimethylpentane, 2,3-	565-59-3	1.52E-01				0.77	0.20		
dimethylphenol, 2,4-	105-67-9		0	8.35E-01		10		0.08	
ethyl mercaptan	75-08-1	6.40E-01				0.85	0.75		
ethylbenzene	100-41-4	5.47E-01		6.20E+01	5.50E-04	8	0.07	**7.75**	0.00
heptanoic acid	111-14-8		0	2.35E+03	4.00E-02	31.885		**73.70**	0.00
hexane	110-54-3	2.44E+00				19.35	0.13		
hexanoic acid	142-62-1 (10051-44-2)		0	6.82E+04	1.40E+01	440		**155.00**	0.03
hydrogen sulphide	7783-06-4	3.89E-02				2	0.02		
iron	7439-89-6	1.11E-04				300	0.00		
isobutyric acid	79-31-2		0	1.04E+04	5.40E+00	808.9		**12.86**	0.01
isophorone	78-59-1		0	2.29E+00		90		0.03	
isovaleric acid	503-74-2		0	2.28E+04	1.10E+01	100		**228.00**	0.11
lead	7439-92-1	3.85E-05				25	0.00		
lindane (gamma BHC)	58-89-9		0	1.20E-02		0.01		**1.20**	
methyl ethyl ketone	78-93-3	8.44E-01		2.76E+01		400	0.00	0.07	
methyl isobutyl ketone	108-10-1	2.33E-01				2300	0.00		
methyl mercaptan	74-93-1	1.58E+00				2.75	0.57		
methylene chloride	75-09-2	4.27E+00		8.03E+02	5.00E-02	100	0.04	**8.03**	0.00

methylheptane, 3-	589-81-1	1.15E+00				0.0235	**49.11**		
naphthalene	91-20-3		0	3.72E+00		7		0.53	
nitrate			0	1.40E+04	2.00E+03	10000		**1.40**	0.20
nitrogen oxides		1.84E+00					#DIV/0!		
nonane	111-84-2	2.89E-01				0.007	**41.25**		
octane	111-65-9	1.67E+00				0.018	**92.95**		
octanoic acid	124-07-2 (1984-06-1)		0	1.10E+04	5.00E-03	12.225		**899.80**	0.00
pentachlorophenol	87-86-5		0	3.16E-01	8.10E-07	0.5		0.63	0.00
pentane	109-66-0	1.37E+00				48.6	0.03		
phenol	108-95-2		0	3.08E+03	5.50E-01	5		**616.00**	0.11
phenyl acetic acid	103-82-2		0	1.63E+03	6.20E-01	55		**29.64**	0.01
propionic acid	79-09-4		0	3.30E+04	1.60E+01	250		**132.00**	0.06
styrene	100-42-5	1.29E-03		3.65E+01		4	0.00	**9.13**	
tetrachloroethane, 1,1,2,2-	79-34-5	6.40E-02		7.92E-01	1.80E-02	70	0.00	0.01	0.00
tetrachloroethylene	127-18-4	1.00E+00		2.97E+00	4.00E-04	50	0.02	0.06	0.00
tetrahydrofuran	109-99-9	2.20E-01		5.72E+00		10800	0.00	0.00	
toluene	108-88-3	2.17E+00		6.75E+02	4.00E-02	0.8	2.72	**843.75**	0.05
trichloroethane, 1,1,1-	71-55-6	3.07E-01		1.10E+01	6.50E-02	10	0.03	**1.10**	0.01
trichloroethane, 1,1,2-	79-00-5	2.54E-02			3.00E-02	800	0.00		0.00
trichloroethylene	79-01-6	5.01E-01		2.34E+01	4.30E-04	20	0.03	**1.17**	0.00
trichlorofluoromethane	75-69-4	1.21E+00		1.69E+00		32.06	0.04	0.05	
trichlorophenol, 2,4,6-	88-06-2		0	3.37E-01	2.20E-06	18		0.02	0.00

vinyl chloride	75-01-4	3.01E-01		1.43E+00	1.71E-01	600	0.00	0.00	0.00
xylene, m-	108-38-3	5.47E-01				2	0.27		
xylene, p-		2.63E-01				30	0.01		
xylene, o-	95-47-6	2.45E-01		7.20E+01	3.10E-03	40	0.01	**1.80**	0.00
xylene, m- & p-				2.14E+02	1.60E-03	2		**107.00**	0.00
zinc	7440-66-6	8.34E-05				20	0.00		

a, There are multiple lines for xylenes because the air chemicals broke it out whereas the leachate and groundwater lists pooled m & p isomers

b, the PWQO for H2S was used for carbonyl sulphide since it converts to H2S on addition to water.

Table C-3.5 Predicted chemical concentrations in surface water from treated leachate.

Landfilling										
Inputs --------------->	CAS no	leachate concentration ug/L	PWQO ug/L	leachate pretreatment removal efficiency (%) default=50	revised conc ug/L	dilution in STP default = 1015.2174	STP removal efficiency %	revised conc ug/L	dilution in receiver x 10	ratio of leachate: PWQO
Chemical										
acetic acid	64-19-7	4.30E+04	395	50	2.15E+04	2.12E+01	50	1.06E+01	1.1	0.003
butyl benzyl phthalate	85-68-7	1.51E+00	0.2	50	7.55E-01	7.44E-04	50	0.00	0	0
butyric acid	107-92-6	5.67E+04	325	50	2.84E+04	2.79E+01	50	1.40E+01	1.4	0.004
chromium VI	7440-47-3	8.80E+01	2	50	4.40E+01	4.33E-02	50	2.20E-02	0	0.001
cresol, o-	95-48-7	3.38E+00	1	99.4	2.03E-02	2.00E-05	99.4	0.00	0	0
cresol, p-	106-44-5	6.00E+02	1	99.4	3.60E+00	3.55E-03	99.4	0.00	0	0
di-n-octylphthalate	117-84-0	2.21E+00	0.2	68.8	6.90E-01	6.79E-04	68.8	0.00	0	0
diethyl phthalate	84-66-2	2.51E+01	0.2	98	5.02E-01	4.94E-04	98	0.00	0	0
dimethyl phthalate	131-11-3	1.31E+00	0.2	96	5.24E-02	5.16E-05	96	0.00	0	0
ethylbenzene	100-41-4	6.20E+01	8	95	3.10E+00	3.05E-03	95	0.00	0	0
hexanoic acid	142-62-1 (10051-44-2)	6.82E+04	440	50	3.41E+04	3.36E+01	50	1.68E+01	1.7	0.004
isovaleric acid	503-74-2	2.28E+04	100	50	1.14E+04	1.12E+01	50	5.62E+00	0.6	0.006
methylene chloride	75-09-2	8.03E+02	100	94.5	4.42E+01	4.35E-02	94.5	2.00E-03	0	0
phenol	108-95-2	3.08E+03	5	99.9	3.08E+00	3.03E-03	99.9	0.00	0	0
phenyl acetic acid	103-82-2	1.63E+03	55	50	8.15E+02	8.03E-01	50	4.01E-01	0	0.001
propionic acid	79-09-4	3.30E+04	250	50	1.65E+04	1.63E+01	50	8.13E+00	0.8	0.003

styrene	100-42-5	3.65E+01	4	99	3.65E-01	3.60E-04	99	0.00	0	0
toluene	108-88-3	6.75E+02	0.8	78	1.49E+02	1.47E-01	78	3.20E-02	0	0.004
heptanoic acid	111-14-8	2.35E+03	32	50	1.18E+03	1.16E+00	50	5.79E-01	0.1	0.002
isobutyric acid	79-31-2	1.04E+04	809	50	5.20E+03	5.12E+00	50	2.56E+00	0.3	0
octanoic acid	124-07-2 (1984-06-1)	1.10E+04	12	50	5.50E+03	5.42E+00	50	2.71E+00	0.3	0.022

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TECHNICAL REPORT – PART “D”

ASSESSMENT OF THE POTENTIAL ECOLOGICAL RISKS ASSOCIATED WITH CHEMICAL EMISSIONS FROM MODERN MUNICIPAL SOLID WASTE INCINERATORS OR LANDFILLS IN ONTARIO - TERRESTRIAL ASPECTS

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Effects of Landfilling and Incineration:
A Comparative Terrestrial Ecological Risk Assessment

D-1 PROBLEM IDENTIFICATION

D-1.1 Definition of Problem

Landfilling and incineration are the primary methods of disposal for municipal solid waste (MSW). An assessment of these disposal methods requires, in part, an understanding of their potential ecological effects. This report is an assessment of the potential impacts of the two disposal methods on the terrestrial (non-human) environment. Potential impacts on human health and aquatic life are covered in the accompanying reports (Reports A, B and C).

D-1.2 Study Objectives

1) To determine the nature and extent of potential adverse impacts on the terrestrial environment resulting from the operation of a state-of-the-art MSW landfilling site.

2) To determine the nature and extent of potential adverse impacts on the terrestrial environment resulting from the operation of a state-of-the-art MSW incinerator handling similar volumes to above landfill operation.

3) To compare the results of the above and to determine if there are significant differences in the potential effects on the terrestrial environment that could result from the choice of disposal method.

D-1.3 Scope of Study

For practical reasons, this study was restricted to examining potential terrestrial effects that could result from the proposed Interim Waste Authority (IWA) landfill Site EE11 in Durham Regional Municipality in southern Ontario, or a similar site, and comparing those results with potential terrestrial effects that could result from a state-of-the-art incinerator or incinerators, of similar total capacity, located at the same site.

Numerous assumptions were made in order to realistically scope the study. First, receptors were chosen which can be said to represent groups of receptors, as it was not possible to consider all potential organisms. Although a complete and thorough assessment would include food web analyses of both northern and southern ecosystems, these more complex analyses were not possible within the timeframe of this study. Receptors were based on professional judgement regarding representative southern Ontario species, as they are likely to be more representative of potential landfill and incineration sites, and, since the Durham site in southern Ontario is the example site of choice. Second, as it is not possible to continue comparing effects indefinitely, a temporal cutoff was made beyond which potential future effects are not considered. The study included the time in which the peak effects are likely to occur for each disposal means.

This study only drew on existing, available information, as well as on additional modelling information that was supplied by Ministry air modelling staff. There was no field or raw data collection component to the study.

D-1.4 Preliminary Identification of Receptors

Since terrestrial effects can occur from direct injury through air, by deposition to the receptor, or by accumulation in soil and uptake or toxicity to the receptor from the soil, the selected receptors included species that account for these pathways. It was necessary to choose receptors and pathways that are common in southern Ontario and which are likely to represent high exposure routes and sensitive species. Since vegetation is the basis of the food supply for animals and since visual injury to vegetation is normally viewed as unacceptable, it was decided to include vegetation as a receptor. No species is specified, as it was necessary to base the potential effects on literature data and experience over a wide range of species, and to choose data which represents effects on common Ontario species in a generic manner.

Grasses and leafy vegetation are common sources of food for a wide range of animals from mice to cattle, and as such it was decided to include them as the basis of the pathways to other receptors. This allows a common base for calculations for wild mammals, domesticated mammals and birds of prey, where bioaccumulation can be an important factor. Since mice and voles are often a major food source for a variety of birds and mammals, it was felt that they should be included as an important receptor in the analysis. This allows an examination of potential effects on both wild mammals and birds through the use of the same food source pathway (leafy vegetation to voles). The final receptors in this sequence were chosen to be hawks (or owls) and fox. These are common species that may be sensitive to contaminants due to the potential for bioaccumulation. It was also felt that cattle should be included as an important domesticated receptor, results for which may be representative of wild ruminants, such as deer. With certain assumptions, the use of these receptors should allow the modelling and assessment of risk to remain manageable while providing considerable useful information.

Organisms that dwell in the soil and are food sources for other species present another route of exposure, which must be examined separately. Due to an earthworm's large intake of soil and its use as a food source by a some bird species, it was felt to be a good representative of soil dwelling organisms for use in this study. Birds are known thought to be sensitive to the effects of persistent chemicals and were therefore included as the final receptor in this pathway. Although robins are a common well-known species of worm-eating bird, the voracious appetite of the woodcock for worms likely makes it a more sensitive indicator of potential environmental stress, hence data from the woodcock was utilized.

The receptor scenarios chosen for assessment in this analyis were as follows;
1) Vegetation (grass and leafy plants)
2) Vegetation → vole → red fox
3) Vegetation → vole → Red-tailed Hawk
4) Earthworm → bird (woodcock, robin)
5) Vegetation (grass or forage) → cattle (which would also represent vegetation → deer)

D-1.5 Conceptual Model of Potential Effects

A conceptual model showing pathways of exposure to contaminants for receptors of concern is presented in Figure D-1.

D-1.6 Determination of Potential Effects to be Examined

There are many potential chemical and physical effects on the terrestrial environment that could be examined in a comparative assessment. These are presented as a matrix in Table D-1.1. It was not possible to assess all possible effects on all receptors. It was decided to assess only those for which a reasonable possibility of finding information or required data exists, and for which the possibility of there being meaningful effects from either landfill or incinerator emissions exists. The decisions regarding what effects are to be examined in this assessment are shown in the matrix. Topics are covered in varying degrees of detail depending on available data and resources. Some are covered in narrative assessments only, while others are assessed quantitatively where possible.

Figure 1: Conceptual Model of Exposure Pathways Used In Assessment

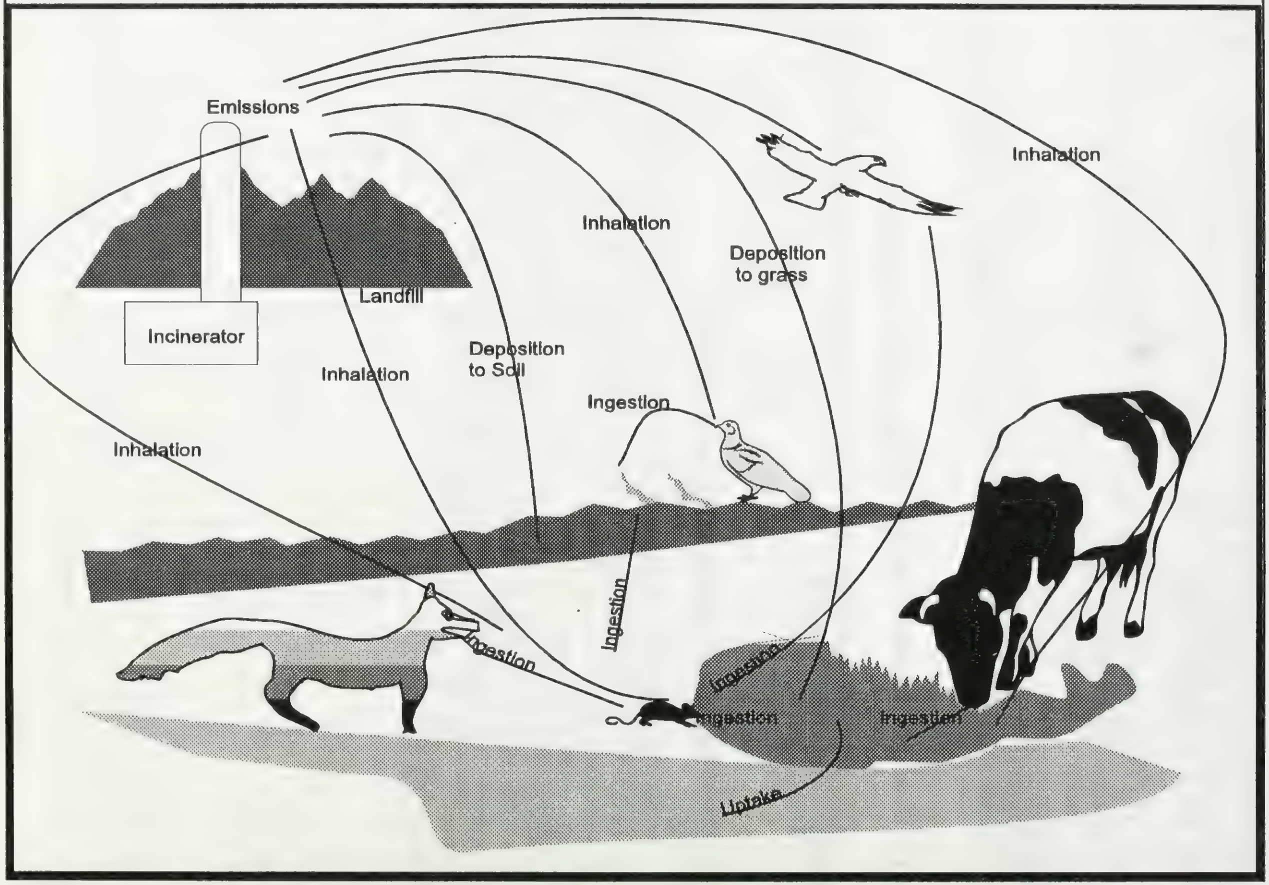

Table D-1.1: Matrix of Potential Terrestrial Effects from Landfills and Incinerators

Contaminant or Stress Type			Animals		Plants	Fungae and Lichen	Bacteria	Protozoa
			Verta-brates	Invert a-brates				
Chemical Contami-nants	Air	gases	Y	Y	Y	N	N	N
		particulate	Y	Y	Y	N	N	N
	Water	surface water	Y	n.a.	Y	N	N	N
		ground water	Y	n.a.	Y	N	N	N
	Soil		Y	Y	Y	N	N	N
Physical	Habitat Loss		Y	Y	Y	N	N	N
	Erosion		Y	Y	Y	N	N	N
	Drainage	air	N	N	Y	N	N	N
		water	N	N	Y	N	N	N
	Water Table		N	N	Y	N	N	N
	Dust		Y	n.a.	Y	N	N	N
	Noise		Y	n.a.	n.a	N	N	N
	Litter		Y	Y	Y	N	N	N
	Odour		N	N	N	N	N	N
	Light		N	N	N	N	N	N
Other	Greenhouse gases and Energy Recovery		Y	N	Y	N	N	N
	Pest species		Y	Y	Y	N	N	N

Y = is covered by either a narrative description or a quantitative or semi-quantitative assessment.
n.a. = not applicable
N = not considered in this assessment.

D-2. SITE CHARACTERIZATION

D-2.1 Background

Since the purpose of this assessment is to compare potential effects of a landfill facility with those of an incinerator, it is important that the site chosen for the comparison is suitable for demonstrating potential differences. The comparative assessment must assume that site characteristics are similar for equity of the comparison to exist. Changes in siting between the two facilities would bring in additional factors that would confound the analysis. For these reasons it was felt that a single site should be chosen that is representative of Southern Ontario and which has a variety of terrestrial receptors nearby. Ideally, there would be considerable information existing about the site and nearby receptors. Since work for the IWA had been completed for three sites, it was felt that one of these sites would be an appropriate basis for constructing the comparative assessment. Since the same site is being used for all three effects assessments (human health, terrestrial effects and aquatic effects), it was felt that the Durham EE11 site was most appropriate since it presents a wide range of terrestrial and aquatic receptors.

D-2.2 Siting and Design Assumptions

For the purposes of this assessment it was assumed that the incinerator is located at the same site as the landfill and that a monofill for ash disposal is operated at the same site. The design of the landfill was assumed to be that of the proposed landfill at the Durham EE11 site, and the design of the incinerator was assumed to be similar to that of the KMS Peel Inc. incinerator. Details are given in other parts of this study as well as in the documents "Detailed Assessment of the Proposed Site EE11, Summary" (Oct. 1994, Interim Waste Authority) and "1994 Emission Testing Report, Peel Resources Recovery Inc. Energy from Waste Facility" (Bovar-Concord Environmental, Feb 1995).

For the purpose of assessing potential effects of chemicals on the terrestrial environment, it was assumed that the receptor of concern, as well as its entire range of feeding activities, is located in the area of the maximum ground-level air concentration of the contaminant of concern.

D-3 RECEPTOR CHARACTERIZATION

The receptor scenarios chosen for assessment in this analyis were as follows;
1) Vegetation (grass and leafy plants)
2) Vegetation → field vole
3) Vegetation → vole → red fox
4) Vegetation → vole → Red-tailed Hawk
5) Vegetation (grass forage) → cattle (which would also represent vegetation → deer)
6) Soil → earthworm → bird (woodcock)

To conduct exposure and hazard assessments for these receptors, it was necessary to determine their characteristics and behaviour which influence the amounts of contaminants taken up. Since this study utilized the CAPCOA (California Air Pollution Control Officers Association) model [CAPCOA, 1993], this section describes and quantifies where possible the information on receptors required by that model. It is noted that the CAPCOA model does not cover the earthworm - bird scenario; hence, additional information was required to assess that route.

The following is a listing of the receptor information required for the exposure assessment. The bracketed letters indicate the codes used in the model.

Vegetation

Vegetation in general was included as a receptor in order to determine if there are species that are sensitive to the chemicals remaining after screening. Therefore, for an assessment of potential effects on vegetation, no specific characteristics were needed, only a search for chemical-specific sensitive species and concentrations at which effects could occur (see the Hazard Assessment Section). Assumptions used in the modelling were as follows;

Interception fraction (IF) - provided in CAPCOA as 0.2 for leafy crops; however, based on the literature a value of 0.39 was selected for leafy crops.
Yield (Y) - seasonal vegetation yield for leafy plants = 2 kg/m^2 [CAPCOA]
Growth period (T) - growth period of 45-90 days was provided in CAPCOA. A 90 day growth period was selected.
Uptake factor (UF2) - chemical specific - taken from CAPCOA tables or determined in the exposure assessment.

Cattle

Inhalation Rate (RR) - 80 m^3/d [CAPCOA)]
Water Ingestion Rate (WI) - 100 kg/d [CAPCOA]
Percentage of water (%SW) ingested from impacted water body - assumed to be 100%.
Feed Ingestion Rate (FIf) - 8 kg/d [CAPCOA)]
Grazing Ingestion Rate (FIg) - 16 kg/d [CAPCOA]
Percentage of diet from grazing (%G) - 67% [CAPCOA]

Percentage of diet as feed (%F) - 33% [CAPCOA]

Soil Ingested (%Sp)- expressed as % of pasture ingested - 5% [CAPCOA]

Soil Ingested (%Sf) - expressed as % of feed other than pasture ingested - 1% [CAPCOA]

Transfer Coefficient (Fi) - fraction of contaminant in inhaled air, ingested water, food and soil, that remains in cattle biomass over its lifetime. This is contaminant specific. Values for dioxin/furans and mercury are provided in Table 1 of CAPCOA. For other contaminants, Fi was determined using the formula: AF*Life/BW, where

- AF - Absorption Factor (assumed to be 100% where no data were available for a chemical.
- Life - average life expectancy of cattle = 548 days (1.5 yr)
- BW - average body weight of cattle = 500 kg [CAPCOA]

Field Vole

Inhalation Rate (RR) - 0.052 m^3/d for an adult male [USEPA, 1993]

Water Ingestion Rate (WI) - 8 g/d [USEPA, 1993]

Percentage of water (%SW) ingested from impacted water body - assumed to be 100%.

Grazing Ingestion Rate (FIg) - 13 g/d [USEPA, 1993]

Percentage of diet from grazing (%G) - assumed to be 100% (grass and leafy plants).

Soil Ingestion Rate (SI) - 0.13 g/d - assumed to be 1% of FIg.

Transfer Coefficient (Fi) - fraction of contaminant in inhaled air, ingested water, vegetation and soil, that remains in vole biomass over its lifetime. Fi was determined using the formula: AF*Life/BW, where

- AF - Absorption Factor (assumed to be 100% where no data were available for a chemical.)
- Life - average life expectancy of field vole = 90 days (3 mo.) [USEPA, 1993]
- BW - average body weight of field vole = 40 g [USEPA, 1993].

Red Fox

Inhalation Rate (RR) - 2 m^3/d for an adult [USEPA, 1993]

Water Ingestion Rate (WI) - 420 g/d [USEPA, 1993]

Percentage of water (%SW) ingested from impacted water body - assumed to be 100%.

Vole Ingestion Rate (FI) - 350 g/d for a non-breeding adult [USEPA, 1993]

Soil Ingestion Rate (SI) - 3.5 g/d - assumed to be 1% of FI.

Transfer Coefficient (Fi) - fraction of inhaled air, ingested water, voles and soil, that remains in fox biomass over its lifetime. Fi was determined using the formula: AF*Life/BW, where

- AF - Absorption Factor (assumed to be 100% where no data was available for a chemical.)
- Life - average life expectancy for red fox = 548 days (1.5 yr) [USEPA, 1993]
- BW - average body weight of red fox (adult male) = 5 kg [USEPA, 1993].

Red-tailed Hawk

Inhalation Rate (RR) - 1.8 m^3/d for an adult bird [Brackenbury, 1986]

Note: based on the literature, the USEPA estimate was too low.

Water Ingestion Rate (WI) - 70 g/d for an adult bird [USEPA, 1993]

Percentage of water (%SW) ingested from impacted water body - assumed to be 100%

Vole Ingestion Rate (FI) - 120 g/d for an adult bird [USEPA, 1993]

Soil Ingestion Rate (SI) - 3.5 g/d - assumed to be 1% of FI.

Transfer Coefficient (Fi) - fraction of contaminant in inhaled air, ingested water, voles and soil, that remains in hawk biomass over its lifetime. Fi was determined using the formula: AF*Life/BW, where

AF - Absorption Factor (assumed to be 100% where no data were available for a chemical.)

Life - average life expectancy for redtail hawk = 3285 days (9yrs) [USEPA, 1993]

BW - average body weight of redtail hawk (adult female) = 1.2 kg [USEPA, 1993].

Worm

Soil Ingestion rate = 7 kg/yr [Barley, 1961]

Woodcock

Worm Ingestion Rate = 150 g/d [Barley, 1961]

Number of days feeding = 180 days

Bird Weight = 0.219 kg (female), 0.176 kg (male) [Dunning, 1984]

For References, see Section D-6.10.

D-4. EXPOSURE ASSESSMENT

D-4.1 Screening of Chemicals

Since reports regarding emissions from landfills and incinerators contain data on hundreds of chemicals, and since is impractical and unnecessary to conduct a complete assessment of the effects of every chemical, techniques were used to screen out chemicals from further consideration where it was apparent that effects would be immeasureable or negligable. The screening system used was as follows:

D-4.1.1 Air Emissions Screening Criteria

1) If stack emission tests indicate that the substance is below detection limits, it will not be assessed.

Rationale: Considerable field experience with terrestrial effects of air emisions indicates that if concentrations are below detection limits in the stack emissions, the substance is highly unlikely to be of concern to the terrestrial environment. If detection limits are too high and the possibility of concern could still exist, the data available is not sufficient to conduct an assessment with any meaningful degree of accuracy.

2) For compounds that co-deposit with particulates, if calculations indicate that maximum rates of deposition would not raise the concentrations in soil (or vegetation) from the mean for Ontario rural parkland background to above the OTR (98th percentile) for rural parkland over the lifetime of the facility, then no further assessment was conducted. A corollary of this is that, if the concentration of the substance on the particulate is below the OTR, then that substance will not be considered, since deposition can not possibly result in a soil number higher than the OTR.

Rationale: At concentrations in soil below the OTR for rural parkland, it is highly unlikely that any meaningful effects would occur. Since these numbers are low-level, background concentrations, ecosystems are exposed to them under natural conditions and any theoretical potential effects are inseparable from those occurring in uncontaminated natural environments. An assumption is made here that the concentrations in the area around the site are likely to be starting at near the median rural parkland values for the province.

3) For Volatile Organic Compounds (VOCs), where long-term deposition to soil is not of potential concern, if calculated concentrations at the point of impingement (POI) are not elevated above analytical detection limits, the compound will not be assessed further. A

check was made for known carcinogenic compounds to assure that this screening does not eliminate chemicals for which a potential for effects could occur below detection limits. The check assumes that if the risk of cancer in humans from the presence of the compound at the calculated POI concentrations is less than 10^{-6}, then concern for non-human species is unwarranted for the purposes of this study. This assumption is reasonable and conservative, since the assumptions used to develop human risk values tend to be very conservative and since population effects in animals at a 10^{-6} risk are likely to be insignificant in comparison to the overall risks in a natural environment. Although the unit risks used in this assessment were developed for humans, much of the data on which they are based were derived from experiments on animals. The same unit risks that were used in the human health risk assessment were therefore used, except in situations where evidence exists that indicates they do not have relevance to other animals.

Rationale: If maximum concentrations are not elevated above analytical detection limits, the likelihood of a potential for terrestrial effects is extremely low and, if effects were theoretically possible for a specific compound, it will not be possible to find any meaningful terrestrial effects since no research work will have been conducted at those concentrations. It would therefore be pointless to conduct a more detailed assessment.

4) For compounds that disperse widely and contribute to acid deposition, if the contribution to total loadings in Ontario is less than 0.1% of those loadings, the compound will not be assessed.

Rationale: Since compounds that disperse widely could have an incremental regional or provincial effect, comparisons must be based on provincial loadings. If they are insignificant compared to current provincial loadings, effects will not be able to be determined.

D-4.1.2 Leachate Emissions

State-of-the-art landfill design, such as that being proposed for the Durham site, includes leachate collection systems within the landfill areas, as well as natural and artificial barriers to prevent or reduce the escape of leachates into natural environment. The data from the assessment of the landfill, as shown in Table D-4.1 indicates that the concentrations of all organic contaminants in the groundwater around the site would not, even under a worst case scenario, begin to approach the effects-based numbers that have been used in the proposed soil clean-up guidelines for the protection of groundwater. The document [IWA, 1994, Appendix QD] indicates that there is no anticipated movement of inorganic constituents through the liners and natural soil layers into the ground water. The major route for a potential effect from leachate on the terrestrial environment would be through either irrigation of crops or for drinking water for animals. Assuming that the modelling conducted for the Durham landfill site is correct, and that the same conclusions can be applied to leachate from a monofill for incinerator ash, these routes of exposure to terrestrial receptors would be anticipated to be minimal. However, they will be assessed, if possible, for those contaminants that remain for detailed examination for other reasons. It is beyond the scope of this study to assess the potential effects of the leachate after treatment at the sewage treatment plant. The chapter on aquatic effects examines these pathways.

Table D-4.1: Comparison of predicted concentrations of organic constituents in the groundwater at the proposed Durham landfill site with Site Clean-up Criteria.

Chemical	Predicted Concentrations in the Groundwater Aquifer (μg/L)		Groundwater Criteria from Table A of Soil Clean-up Guidelines (μg/L)
	Anticipated	Worst Case	
Methylene chloride	8.8E-9	0.05	50
1,1-Dichloroethylene	2.2E-6	0.042	0.66
Vinyl Chloride	5.5E-5	0.17	0.5
Benzene	8.8E-8	0.026	5
1,1,2,2-tetrachloroethane	1.8E-12	0.018	1
Bromochloromethane	2.1E-7	2.8E-4	5
1,2-Dichloroethane	2.2E-5	0.028	5
Pentachlorophenol	1E-11	8.1E-7	30
1,4-Dichlorobenzene	1.6E-9	6.8E-3	1
Chloroform	8.7E-7	0.027	5
Phenol	1.3E-6	0.55	4200
2,4,6 Trichlorophenol	5.5E-8	2.2E-6	2
Toluene	1.5E-11	0.04	24
Trichloroethylene	9.6E-19	4.3E-4	50
Bromodichloromethane	7.5E-7	0.031	5
Tetrachloroethylene	2.2E-18	4.0E-4	5
1,1,1-Trichloroethane	5.9E-10	0.065	200
1,1,2-Trichloroethane	5.5E-10	0.03	5
1,1-Dichloroethane	0.0091	4.8	70
cis 1,2-Dichloroethylene	0.01	0.75	70
Ethylbenzene	1E-23	5.5E-4	2.4
m&p Xylenes	1E-22	.0016	300
o-Xylenes	8.3E-24	.0031	300

D-4.1.3 Results of Screening

D-4.1.3.1 Screening for Chemicals From Incineration

1) Detection Limits in Stack

None of the inorganics in the test list occurred in the stack emissions at levels below detection limits. This level of screening resulted in the elimination of 53 organic chemicals from further examination. Organics that remain after screening out due to concentrations being below detection limits in all six 1994 stack emission tests at KMS Peel Inc.,are as follows;

1,2-Dichlorobenzene
1,3-Dichlorobenzene
1,4-Dichlorobenzene
1,2,3-Trichlorobenzene
1,2,4-Trichlorobenzene
1,2,3,4-Tetrachlorobenzene
1,2,3,5 and 1,2,4,5-Tetrachlorobenzene
Pentachlorobenzene
2,3,4-Trichlorophenol
2,3,4,6-Tetrachlorophenol
Pentachlorophenol
Tetralin
2-Methylnaphthalene
1-Methylnaphthalene
Biphenyl
o-Terphenyl
Naphthalene
Acenaphthalene
Acenaphthene
Fluorene
Phenanthrene
Anthracene
Fluoranthene
Pyrene
Dioxins and Furans (all congeners)

The documentation for the KMS Peel Inc.,incinerator does not show any results for Benzene, Toluene, Ethylbenzene, Xylene, (BTEXs) or for chlorinated aliphatics. Data from the Hartford, Connecticut MSW incinerator were available and covered some of these chemicals. The data indicate that some of these chemicals are emitted in significant quantities and should be included. In addition, the testing of the Hartford facility covered different PAHs than did that at the KMS Peel Inc.,facility. On this basis, the following chemicals were included in the assessment.

Benzene
Chloromethane
Bromomethane
Vinyl Chloride
Toluene
Dichlorodifluoromethane
Trichlorofluoromethane
Xylenes
Dibromoethane
1,1-Dichloroethene
1,1,1-Trichloroethane
Trichloroethene
Tetrachloroethene
Ethylbenzene
2-Methyl-fluorene
1-Methyl-pyrene

For the next screening, the maximum 24-hour ground level concentrations resulting from using the worst case emissions from two years (1993 and 1994) of monitoring data from Peel, combined with data from the Hartford facility for those chemicals not monitored at Peel, were utilized. A modelling excercise was conducted by the Science and Technology Branch to determine potential ground level concentrations (GLCs) from a similar incinerator if it were to be placed at the Durham site. Calculations were made to determine potential ground level concentrations at hundreds of points in a large grid around the facility. For the screening of chemicals present at above detection limits in the stack emissions, a dispersion factor representing the maximum 24-hour GLC was used. These data are shown in Table D-4.2.

Table D-4.2: Incinerator emissions data for chemicals measured at above detection limits

Parameter	Uncorrected Emission rate (μg/s)	Corrected Emission rate[1] (μg/s)	Corrected 24 hr Max GLC ($\mu g/m^3$)	Corrected Annual Average max GLC ($\mu g/m^3$)	Total Emissions (kg/yr)
Flue gas flow rate (m^3/s)	18.5	30			
Particulate emissions		117740	0.20	0.0088	3709
Sulphur Dioxide		1051143	1.79	0.0788	195828.45
Nitrogen Oxides		19044242	32.38	1.42	1201310.83
Hydrogen Chloride		790781	1.34	0.025	42081.14
Aluminum	752	1093.09	1.86E-03	8.20E-05	34.48
Antimony	29	42.15	7.17E-05	3.16E-06	1.33
Arsenic	15	21.80	3.71E-05	1.64E-06	0.69
Barium	40	58.14	9.88E-05	4.36E-06	1.83
Beryllium	6.3	9.16	1.56E-05	6.87E-07	0.29
Boron	2026	2944.95	5.01E-03	2.21E-04	92.88
Cadmium	49	71.23	1.21E-04	5.34E-06	2.25
Calcium	36389	52894.28	8.99E-02	3.97E-03	1668.29
Chromium	1488	2162.93	3.68E-03	1.62E-04	68.22
Cobalt	109.6	159.31	2.71E-04	1.19E-05	5.02
Copper	165.7	240.86	4.09E-04	1.81E-05	7.60
Iron	4431	6440.81	1.09E-02	4.83E-04	203.14
Lead	4941	7182.13	1.22E-02	5.39E-04	226.52
Lithium	87.8	127.62	2.17E-04	9.57E-06	4.03

Parameter	Uncorrected Emission rate ($\mu g/s$)	Corrected Emission rate[1] ($\mu g/s$)	Corrected 24 hr Max GLC ($\mu g/m^3$)	Corrected Annual Average max GLC ($\mu g/m^3$)	Total Emissions (kg/yr)
Magnesium	688	1000.06	1.70E-03	7.50E-05	31.54
Manganese	202	293.62	4.99E-04	2.20E-05	9.26
Mercury	8690	12631.60	2.15E-02	9.47E-04	398.40
Molybdenum	80.6	117.16	1.99E-04	8.79E-06	3.70
Nickel	1648	2395.50	4.07E-03	1.80E-04	75.55
Phosphorus	871	1266.07	2.15E-03	9.50E-05	39.93
Selenium	36	52.33	8.90E-05	3.92E-06	1.65
Silicon	816782	1187256.94	2.02E+00	8.90E-02	37446.08
Silver	63.4	92.16	1.57E-04	6.91E-06	2.91
Sodium	14579	21191.72	3.60E-02	1.59E-03	668.39
Strontium	21	30.53	5.19E-05	2.29E-06	0.96
Tin	332.9	483.90	8.23E-04	3.63E-05	15.26
Titanium	72	104.66	1.78E-04	7.85E-06	3.30
Vanadium	22	31.98	5.44E-05	2.40E-06	1.01
Zinc	3775	5487.26	9.33E-03	4.12E-04	173.07
Fluoride	14.6	21.22	3.61E-05	1.59E-06	0.67
1,2-Dichlorobenzene	38.7	56.25	9.56E-05	4.22E-06	1.77
1,3-Dichlorobenzene	12.2	17.73	3.01E-05	1.33E-06	0.56
1,4-Dichlorobenzene	16.3	23.69	4.03E-05	1.78E-06	0.75
Total Trichlorobenzenes	18.2	26.46	4.50E-05	1.98E-06	0.83
Total Tetrachlorobenzenes	4.2	6.11	1.04E-05	4.58E-07	0.19
Pentachlorobenzene	2.56	3.72	6.33E-06	2.79E-07	0.12
Total Trichlorophenols	18.2	26.46	4.50E-05	1.98E-06	0.83
2,3,4,6-Tetrachlorophenol	3.29	4.78	8.13E-06	3.59E-07	0.15
Pentachlorophenol	3.9	5.67	9.64E-06	4.25E-07	0.18
Tetralin	9.43	13.71	2.33E-05	1.03E-06	0.43
Biphenyl	56.43	82.03	1.39E-04	6.15E-06	2.59
o-Terphenyl	1.55	2.25	3.83E-06	1.69E-07	0.07
2-Methylnaphthalene	5.2	7.56	1.28E-05	5.67E-07	0.24
1-Methylnaphthalene	1.86	2.70	4.60E-06	2.03E-07	0.09
2-Methylfluorene	289	420.08	7.14E-04	3.15E-05	13.25

Parameter	Uncorrected Emission rate (μg/s)	Corrected Emission rate[1] (μg/s)	Corrected 24 hr Max GLC (μg/m^3)	Corrected Annual Average max GLC (μg/m^3)	Total Emissions (kg/yr)
1-Methylpyrene	0.32	0.47	7.91E-07	3.49E-08	0.01
Naphthalene	26.52	38.55	6.55E-05	2.89E-06	1.22
Acenaphthylene	3.9	5.67	9.64E-06	4.25E-07	0.18
Acenaphthene	1.8	2.62	4.45E-06	1.96E-07	0.08
Fluorene	2.2	3.20	5.44E-06	2.40E-07	0.10
Phenanthrene	11.6	16.86	2.87E-05	1.26E-06	0.53
Fluoranthene	13.2	19.19	3.26E-05	1.44E-06	0.61
Pyrene	7.2	10.47	1.78E-05	7.85E-07	0.33
Dioxins/Furans (Total TEQ)	0.0056	0.01	1.38E-08	6.11E-10	0.00
Chloromethane	2973	4321.49	7.35E-03	3.24E-04	136.30
Bromomethane	681	989.89	1.68E-03	7.42E-05	31.22
Dibromoethane	8.2	11.92	2.03E-05	8.94E-07	0.38
Dichlorodifluoromethane	1647	2394.04	4.07E-03	1.80E-04	75.51
Trichlorofluoromethane	3258	4735.76	8.05E-03	3.55E-04	149.37
Vinyl Chloride	825	1199.20	2.04E-03	8.99E-05	37.82
1,1-Dichloroethene	10.7	15.55	2.64E-05	1.17E-06	0.49
Trichloroethene	9.3	13.52	2.30E-05	1.01E-06	0.43
1,1,1-Trichloroethane	27	39.25	6.67E-05	2.94E-06	1.24
Tetrachloroethene	42.8	62.21	1.06E-04	4.67E-06	1.96
Benzene	51366	74664.53	1.27E-01	5.60E-03	2354.92
Toluene	951	1382.35	2.35E-03	1.04E-04	43.60
Ethylbenzene	19.6	28.49	4.84E-05	2.14E-06	0.90
Total xylenes	11427	16610.04	2.82E-02	1.25E-03	523.88

Peel/Hartford MSW Incinerator - Maxima from 1993 & 1994 Peel tests OR 3X average of 1 boiler test from Hartford

[1] All values corrected for stack temperature to ground level temperature (25 Celsius)

TEQ - 2,3,7,8-TCDD toxic equivalents calculated using International toxic equivalent factors

2) Comparison of Accumulated Deposition with Background

This screening method was applied to inorganics, PAHs and Dioxins/furans that remained after the initial screening. VOCs were not considered as they disperse very widely and do not readily move into soil from the air. The comparison of calculated maximum soil concentrations (mean background plus worst case accumulation over 20 years) with the Ontario Typical Range 98th% value (OTR_{98}) resulted in the elimination of all the inorganics except mercury, lithium, silicon, and tin. The latter three were not excluded by this screening as there are no OTRs for them. Adriano [1986] indicates that a common range for tin in surface soils is from about 1 to 10 μg/g, and that the mean and range in the USA are 0.89 μg/g and from 0.1 to 10 μg/g respectively. The calculated worst case addition of 0.0208 μg/g due to 20 years of incinerator operation is insignificant in comparison, and hence tin need not be considered for further assessment. Dragun [1988] gives the typical range of soil concentrations of lithium to be from 7 to 200 μg/g and the extreme limits to be from 1 - 3000 μg/g. A comparison of the calculated worst case deposition scenario over 20 years of 0.0055 μg/g added to the soil with these ranges indicates clearly that deposition of lithium from incinerator emissions was excluded from further assessment. Since silica is a major constituent of soil at 230,000 - 350,000 μg/g in natural soils [Dragun, 1988], the 51 μg/g added by incinerator emissions is also clearly insignificant with respect to background soil concentration. If concern were to exist with silicon, it would be from the perspective of particulate emissions and their effects, not as silicon per se. This aspect is covered later under the effects of dust (Section 6.4).

Although the screening as conducted does not show that 20 years of deposition of dioxins and furans would result in exceedences of upper background concentrations, it was felt that, due to the high degree of public interest in these substances, especially with respect to incinerator emissions, their effects on terrestrial receptors should be examined more thoroughly, and they were retained for detailed assessment.

Of the chlorinated phenols, 2,3,4-trichlorophenol, 2,3,4,6-tetrachlorophenol, and pentachlorophenol were detected in the stack emissions testing for the KMS Peel Inc.,facility. Although these compounds are not highly volatile, it is unlikely that they would all remain bound to particulate matter in the emissions. In a deposition scenario, therefore, the worst case would be that all of these compounds deposited with particulates. If this were the case, calculations indicate that total chlorinated phenols would not exceed 0.0033 μg/g (plus original background) in the soil after 20 years, assuming no degradation and that all the individual chlorinated phenolics were present in the stack emissions at detection limits. Since the MOEE method detection limits (MDLs) in soil are approximately 0.10 μ/g for each compound, these compounds could not be present at more than one thirtieth of the current MOEE MDL after 20 years, and it is not likely that they would be of concern with respect to the terrestrial environment. As such, they were not considered for further assessment.

Tetralin, biphenyl o-terphenyl and 1-methylpyrene were compounds detected in stack emissions for which no OTR_{98} data existed for comparison. For these four compounds, the calculated additions to soil over the 20 year period are 0.0006, 0.0035, 0.0001 and 0.00002 μg/g respectively. The lowest practical instrument detection limits for these compounds are about 0.006 μg/g with MDLs considerably higher. Since the total addition of these chemicals to the soil over 20 years, assuming worst case deposition rates and no degradation, would not be detectable, they were not considered for further assessment. 2-methylfluorene however, was retained for detailed assessment since the maximum soil addition of 0.018 μg/g is similar to method detection limits, and no OTR exists.

Table D-4.3: Comparison of deposition from incinerator over 20 years with background concentrations in soil

Parameter	OTR_{50} rural (μg/g)	OTR_{98} rural (μg/g)	Max GLC (μg/m^3)	Deposition (μg/m2/yr)	Deposition in 5cm soil (μg/g)	Maximum Soil Conc (μg/g)	Deposition /OTR_{98} Ratio
Aluminum	16000	30000	1.86E-03	117.20	4.69E-02	1.60E+04	0.0000016
Antimony	0.15	0.43	7.17E-05	4.52	1.81E-03	1.52E-01	0.0042
Arsenic	4	11	3.71E-05	2.34	9.35E-04	4.00E+00	0.000085
Barium	70	160	9.88E-05	6.23	2.49E-03	7.00E+01	0.000016
Beryllium	0.45	1.1	1.56E-05	0.98	3.93E-04	4.50E-01	0.00036
Boron	14	30	5.01E-03	315.76	1.26E-01	1.41E+01	0.0042
Cadmium	0.25	0.71	1.21E-04	7.64	3.05E-03	2.53E-01	0.0043
Calcium	10000	55000	8.99E-02	5671.45	2.27E+00	1.00E+04	0.000041
Chromium	25	58	3.68E-03	231.91	9.28E-02	2.51E+01	0.0016
Cobalt	7.5	16	2.71E-04	17.08	6.83E-03	7.51E+00	0.00043
Copper	16	41	4.09E-04	25.83	1.03E-02	1.60E+01	0.00025
Iron	20000	35000	1.09E-02	690.60	2.76E-01	2.00E+04	0.0000079
Lead	22	45	1.22E-02	770.09	3.08E-01	2.23E+01	0.0068
Lithium			2.17E-04	13.68	5.47E-03	5.47E-03	
Magnesium	6000	20000	1.70E-03	107.23	4.29E-02	6.00E+03	0.0000021
Manganese	900	1300	4.99E-04	31.48	1.26E-02	9.00E+02	0.0000097
Molybdenum	0.2	1	1.99E-04	12.56	5.02E-03	2.05E-01	0.0050
Nickel	13	38	4.07E-03	256.85	1.03E-01	1.31E+01	0.0027
Phosphorus	540	900	2.15E-03	135.75	5.43E-02	5.40E+02	0.000060
Selenium	0.4	0.93	8.90E-05	5.61	2.24E-03	4.02E-01	0.0024
Silicon			2.02E+00	127300.54	5.09E-01	5.09E+01	
Silver	0.12	0.27	1.57E-04	9.88	3.95E-03	1.24E-01	0.015
Sodium	225	660	3.60E-02	2272.23	9.09E-01	2.26E+02	0.0014
Strontium	28	64	5.19E-05	3.27	1.31E-03	2.80E+01	0.00002

Parameter	OTR_{50} rural (μg/g)	OTR_{98} rural (μg/g)	Max GLC ($\mu g/m^3$)	Deposition (μg/m2/yr)	Deposition in 5cm soil (μg/g)	Maximum Soil Conc (μg/g)	Deposition /OTR_{98} Ratio
Tin			8.23E-04	51.88	2.08E-02	2.08E-02	
Titanium	3200	5200	1.78E-04	11.22	4.49E-03	3.20E+03	0.0000009
Vanadium	40	77	5.44E-05	3.43	1.37E-03	4.00E+01	0.000018
Zinc	60	120	9.33E-03	588.36	2.35E-01	6.02E+01	0.0020
Fluoride	18	61	3.61E-05	2.28	9.10E-04	1.80E+01	0.000015
Mercury	0.06	0.13	2.15E-02	1354.39	5.42E-01	6.02E-01	4.2
Total Dichlorophenols			7.12E-05	4.49	1.80E-03	1.80E-03	
Total Trichlorophenols			4.50E-05	2.84	1.13E-03	1.13E-03	
2,3,4,6-Tetrachlorophenol		0.014	8.13E-06	0.51	2.05E-04	2.05E-04	0.015
Pentachlorophenol		0.014	9.64E-06	0.61	2.43E-04	2.43E-04	0.017
Tetralin			2.33E-05	1.47	5.88E-04	5.88E-04	
Biphenyl			1.39E-04	8.79	3.52E-03	3.52E-03	
o-Terphenyl			3.83E-06	0.24	9.66E-05	9.66E-05	
2-Methylnaphthalene		0.003	1.28E-05	0.81	3.24E-04	3.24E-04	0.11
1-Methylnaphthalene		0.006	4.60E-06	0.29	1.16E-04	1.16E-04	0.019
2-Methylfluorene			7.14E-04	45.04	1.80E-02	1.80E-02	
1-Methylpyrene			7.91E-07	0.05	1.99E-05	1.99E-05	
Naphthalene		0.006	6.55E-05	4.13	1.65E-03	1.65E-03	0.28
Acenaphthylene	0.006	0.03	9.64E-06	0.61	2.43E-04	6.24E-03	0.0081
Acenaphthene		0.006	4.45E-06	0.28	1.12E-04	1.12E-04	0.019
Fluorene		0.009	5.44E-06	0.34	1.37E-04	1.37E-04	0.015
Phenanthrene		0.092	2.87E-05	1.81	7.23E-04	7.23E-04	0.0079
Fluoranthene		0.14	3.26E-05	2.06	8.23E-04	8.23E-04	0.0059
Pyrene		0.11	1.78E-05	1.12	4.49E-04	4.49E-04	0.0041
Dioxins/furans (TotalTEQ)	0.0000016	0.0000048	1.38E-08	8.73E-04	3.49E-07	1.95E-06	0.073

Since the possibility exists that concentrations in air could still be of concern even though deposition rates have ruled out the possibilities for concern due to accumulation in the soil, the calculated GLC values were compared to existing point of impingement (POI) ambient air quality criteria (AAQC) as a double check to assure that no significant oversights occurred in this screening. For all measured contaminants, with the exception of silica, which will be considered later under dust, for which AAQCs exist, the calculated GLC values were orders of magnitude lower than existing AAQC, hence, there is no apparent need to reconsider compounds screened out using the deposition to soil scenario.

Table D-4.4: Comparison of maximum ground level concentrations with ambient air quality criteria

Parameter	Corrected 24-hr Max GLC ($\mu g/m^3$)	AAQC 24 hr ($\mu g/m^3$)	AAQC /max GLC
Aluminum	0.0019	120	0.000016
Antimony	0.000072	25	0.0000029
Arsenic	0.000037	0.3	0.00012
Barium	0.000099	10	0.0000099
Beryllium	0.000016	0.01	0.00156
Boron	0.0050	120	0.000042
Cadmium	0.00012	2	0.000060
Chromium	0.0037	1.5	0.0025
Cobalt	0.00027	50	0.0000054
Copper	0.00041	75	0.0000055
Lead	0.012	5	0.0024
Lithium	0.00022	20	0.000011
Manganese	0.0005	100	0.000005
Mercury	0.021	5	0.0043
Molybdenum	0.0002	120	0.0000017
Nickel	0.0040	2	0.0020362
Selenium	0.000089	10	0.0000089
Silicon	2.0	5	0.40
Silver	0.00016	3	0.000052
Strontium	0.000052	120	0.0000004
Tin	0.00082	10	0.000082
Titanium	0.00018	120	0.0000015
Vanadium	0.000054	2	0.000027
Zinc	0.0093	120	0.000078

Parameter	Corrected 24-hr Max GLC ($\mu g/m^3$)	AAQC 24 hr ($\mu g/m^3$)	AAQC /max GLC
Fluoride	0.000036	0.86	0.000042
Pentachlorophenol	0.0000096	20	0.0000005
Biphenyl	0.00014	33	0.0000042
Naphthalene	0.000065	22.5	0.0000029
Peel/Hartford MSW Incinerator - Maxima from 1993 & 1994 Peel tests OR 3X average of 1 boiler test from Hartford All values corrected for stack temperature to ground level (25^{o} Celsius)			

As a result of this screening procedure, the following chemicals remain for further assessment:

Mercury
1,2-Dichlorobenzene
1,3-Dichlorobenzene
1,4-Dichlorobenzene
1,2,3-Trichlorobenzene
1,2,4-Trichlorobenzene
1,2,3,4-Tetrachlorobenzene
1,2,3,5 and 1,2,4,5-Tetrachlorobenzene
Pentachlorobenzene
Benzene
Ethylbenzene
Chloromethane
Bromomethane
1,1-Dichloroethene
Trichloroethene
Vinyl Chloride
Toluene
Xylene
Dichlorodifluoromethane
Trichlorofluoromethane
2-Methyl-fluorene
SO,
NO,
HCl
Dioxins and furans
Dibromoethane
1,1,1-Trichloroethane
Tetrachloroethene

3) Comparison of Air Concentrations with Analytical Detection Limits for VOCs

The following table shows the calculated GLCs and the analytical method detection limits (MDLs) for the remaining VOCs.

Table D-4.5: Estimated maximum ground level concentrations and analytical detection limits - incinerator

Compound	Maximum GLC ($\mu g/m^3$)	MDL ($\mu g/m^3$)
1,2-Dichlorobenzene	0.0000956	1.0
1,3-Dichlorobenzene	0.0000301	0.5
1,4-Dichlorobenzene	0.0000403	0.5
Total trichlorobenzenes	0.0000450	≥0.1
Total tetrachlorobenzenes	0.0000104	≥0.1
Pentachlorobenzene	0.0000063	≥0.1
Hexachlorobenzene	0.0000007	≥0.1
Benzene	0.127	0.2
Toluene	0.00235	0.5
Xylenes	0.0282	1
Toluene	0.00235	0.5
Ethylbenzene	0.000048	0.5
Chloromethane	0.00735	≥0.1
Bromomethane	0.00168	≥0.1
Dibromoethane	0.0000203	≥0.1
Vinyl Chloride	0.00204	≥0.1
Dichlorodifluoromethane	0.00407	≥0.1
Trichlorofluoromethane	0.00805	≥0.1
1,1-dichloroethene	0.000026	≥0.1
1,1,1-trichloroethane	0.0000667	0.1
trichloroethene	0.000023	0.25
tetrachloroethene	0.00011	0.25
MDLs were obtained from the Document "Guidance on Sampling and Analytical Methods for Use at Contaminated Sites in Ontario" or, where the "≥" sign appears, as a minimum value based on MOEE Laboratory Services Branch experience and personal communications with MOEE laboratory staff.		

Of the above compounds where the calculated GLC is below detection limits, benzene, vinyl chloride and chloromethane require special consideration prior to screening out, as they are known human carcinogens. The calculated GLC for vinyl chloride is 0.00204 $\mu g/m^3$, and the unit risk is 8.4×10^{-5} (per $\mu g/m^3$), giving a human risk of 1.7×10^{-7}. Using the assumptions discussed earlier, this indicates that it is unwarranted to examine vinyl chloride in further detail. Similarly, chloromethane with a calculated GLC of .00735 $\mu g/m^3$ and a unit risk of 1.8×10^{-6} (giving a risk of 1.3×10^{-8}) would also be screened out. For benzene, using a unit risk of 8.3×10^{-6}, the inhalation risk would be 1.05×10^{-6}, and accordingly, was added back to the list.

This screening method indicates that, of the above chemicals, only benzene would appear to have potential for concern.

As a result of this screening procedure the following chemicals remain for further assessment;

Mercury
Benzene
2-Methyl-fluorene
SO_2
NO_x
HCl
Dioxins and furans

5) Contribution to Total Loadings in Ontario

Emission rates for the KMS Peel Inc.,incinerator running at approximately 400 tonnes per day are 1.05 g/s of sulphur dioxide (SO_2) and 19 g/s of nitrogen oxides (NO_x) (226 g/tonne and 4100 g/tonne respectively). This corresponds to 33 tonnes of SO_2 and 600 tonnes of NO_x per year, assuming constant operation of all four burners (i.e. 146,000 tonnes of MSW). At these rates, the incineration of 6.6 million tonnes of MSW would produce approximately 1,490 tonnes of SO_2 and 27000 tonnes of NO_x emissions into the air. If this occurred over the 20-year lifespan of a facility, then 74 tonnes and 1,350 tonnes /yr respectively would be emitted. Total Ontario emissions for 1993 were 840,000 tonnes of SO_2 and 545,000 tonnes of NO_x [Simon Wong, OMEE, personal communication]. The annual emissions of an incinerator handling 6.6 million tonnes of MSW over 20 years would therefore represent approximately 0.01% of Ontario's SO_2 emissions and 0.25% of Ontario's NO_x emissions. Since the screening criteria was set at 0.1% of Ontario emissions, NO_x emissions from incinerators are sufficiently high to warrant further investigation.

Final List of Chemicals for Assessment for Incinerator Emissions

As a result of the screening procedures, the following chemicals remain for further assessment;

Mercury
Benzene
2-Methyl-fluorene
NO_x
HCl
Dioxins and furans

D-4.1.3.2 Screening for Chemicals from Landfills

Leachates

As described in Section D-4.1.2, the data from the environmental assessment of the Durham landfill site indicate that the concentrations of chemicals in the groundwater aquifer most likely to be affected by the landfill, will not exceed the effects-based numbers that were derived for the protection of aquifers in the proposed guidelines for contaminated sites in Ontario. Since these numbers account for human-health effects, aquatic effects, and volatilization to indoor air, then, assuming that the modelling for the Durham site is appropriate, exposure of terrestrial receptors to landfill emissions of the chemicals modelled through groundwater or surface waters by ingestion or irrigation would be unlikely to be of concern in itself, and need not be examined further. Chemicals which were not modelled or which are not covered in Table A of the proposed contaminated sites guidelines will only be examined where they are of concern from another pathway (i.e. air) or where the health or surface water assessments indicate the potential for concern. The assumptions used here are that chemicals that are not a concern for human-health in drinking water or for aquatic species in surface waters are unlikely to be of concern in water supplies for terrestrial animals or plants.

Air Emissions.

The list of candidate chemicals was taken from the combined list of chemicals detected in the vents at the Brittania landfill and other landfills as presented in Table QD 19 of the document "Appendix QD, Character and Estimation of Chemical Emission, Detailed Assessment of the Proposed Site EE11" [Cantox, 1994]. For initial screening, concentrations in air at the receptor were calculated using an averaged factor between the calculated values in that document and the vent concentrations (i.e. values in Table QD23 divided by values in Table QD19, which was 1.6 x 10^{-6}).

Compounds or elements that are associated with particulate emissions are not considered to be of concern at landfill sites, since particulates are not evolved from the landfill in the gases. These materials tend to remain in the soil at the site. Although there is often a concern regarding dust, which will be addressed in this study, the IWA assessments indicated that the dust is normally little different than dust associated with normal soil. Hence, this can be treated as a problem of dust and not of the chemicals contained on the dust that are attributable to the landfill. Thus, emissions relating to chemicals on particulates were not be considered further in this assessment. The validity of this approach is borne out by the calculated values for concentrations of inorganic constituents at the receptor from the Durham IWA report. These values tend to be three orders of magnitude lower than existing detection limits in air.

The chemicals remaining were then screened based on the calculated maximum 24-hour GLCs, exceeding detection limits for standard air sampling, as was done for incinerator chemicals.

The list of chemicals and method detection limits used is listed in Table D-4.6 below.

Table D-4.6: Ground level air concentrations and method detection limits - Landfill

Chemical	Maximum 24-hr air Concentration at Receptor ($\mu g/m^3$)	MDL ($\mu g/m^3$)
Butan-2-ol	1.936	
Dichlorodifluoromethane	1.888	
Methylene chloride	1.468	0.5
Hexane	0.8432	
Toluene	0.7504	0.5
Decane	0.664	
Butane	0.6496	
Nitrogen oxides	0.6352	
1,1-Dichloroethane	0.6226	0.1
Octane	0.5776	
Chlorodifluoromethane	0.4832	
Pentane	0.4736	
Trichlorofluoromethane	0.416	
3-Methylheptane	0.3984	
Tetrachloroethylene	0.3456	0.25
Carbonyl sulphide	0.3312	
Methyl mercaptan	0.3276	
Dichlorofluoromethane	0.296	
MEK (Methyl ethyl ketone)	0.2912	
Acetone	0.2624	
Bromodichloromethane	0.2208	0.25
Ethyl mercaptan	0.2191	
Ethylbenzene	0.1888	0.5
m-Xylene	0.1888	1
Trichloroethylene	0.1715	0.25
Cyclohexane	0.1549	
Dimethylsulphide	0.1394	
1,1,1-Trichloroethane	0.1061	0.1

Chemical	Maximum 24-hr air Concentration at Receptor ($\mu g/m^3$)	MDL ($\mu g/m^3$)
Vinyl chloride	0.1035	0.25
Nonane	0.1011	
p-Xylene	0.09072	1
o-Xylene	0.08448	1
MIK (Methyl isobutyl ketone)	0.08032	
THF (Tetrahydrofuran)	0.07584	
2,3-Dimethylpentane	0.05264	
1,4-Dichlorobenzene	0.04352	0.5
Carbon disulphide	0.03504	
Benzene	0.03488	0.2
Butylacetate	0.02752	
Hydrogen sulfide	0.01344	
1,2-Dichloropropane	0.012624	0.2
1,1,2-Trichloroethane	0.008704	0.25
1,2-Dichloroethane	0.00712	0.1
1,2-Dichloroethylene	0.005664	0.1
Chloroform	0.00304	0.1
1,3-Dichlorobenzene	0.00184	0.5
Dimethyldisulphide	0.001232	
1,2-Dibromoethane	0.000864	>0.1
Chlorobenzene	0.000816	0.25
Carbon tetrachloride	0.0008	0.1
1,2-Dichlorobenzene	0.00048	1
Styrene	0.000448	1
Iron	0.0000384	0.05
Zinc	0.0000288	0.025
Copper	0.0000176	0.01
Lead	0.00001328	0.01
Chromium	0.0000048	0.01
Cadmium	0.00000014	0.01
1,1-Dichloroethylene	0.0476	0.1
1,1,2,2-Tetrachloroethane	0.022	0.5

Chemical	Maximum 24-hr air Concentration at Receptor ($\mu g/m^3$)	MDL ($\mu g/m^3$)
Chloroethane	0.0498	>0.1
Chloromethane	0.0317	>0.1
Acrilonitrile	0.00135	

The list was checked for human carcinogens as described in Section D-4.1.1 (3). Although calculated maximum GLCs for vinyl chloride, 1,1-dichloroethylene and 1,1,2,2-tetrachloroethane and bromodichloromethane are below the method detection limits for ambient air, each of these chemicals could pose a risk of greater than 10^{-6} (to humans), as shown below, and therefore are retained for further assessment.

Table D-4.7: Human risk levels of carcinogens near landfill

	Unit Risk (ug/m3)E-1	Reference	Concentration vent (a) ug/m3	Maximum 24-h concentration at receptor (b) ug/m3	Risk
Clorinated Organics					
1,1-Dichloroethane (c)	2.60E-05	IRIS, 1994	38912	0.623	1.62E-5
Vinyl chloride	8.40E-05	HEAST, 1992	6467	0.103	8.7E-6
Bromodichloromethane (e)	2.00E-05	IRIS, 1994	13803	0.22	4.4E-6
1,1-Dichloroethylene	5.00E-05	IRIS, 1994	2975	0.0476	2.4E-6
1,1,2,2-Tetrachloroethane	5.80E-05	IRIS, 1994	1374	0.022	1.3E-6
Methylene chloride	4.70E-07	IRIS, 1994	91780	1.468	6.9E-7
Trichloroethylene	2.00E-06	CAPCOA, 1993	10716	0.171	3.4E-7
1,2-Dichloroethane	2.60E-05	IRIS, 1994	445	0.00712	1.9E-7
,1,2-Trichloroethane	1.60E-05	IRIS, 1994	544	0.00870	1.4E-7
Chloroethane (d)	1.80E-06	IWA, 1994	3110	0.04986	8.95E-08
Chloroform	2.30E-05	IRIS, 1994	190	0.00304	7.0E-08
Chloromethane	1.80E-06	HEAST, 1992	1982	0.0317	5.7E-08

Carbon tetrachloride	1.50E-05	IRIS, 1994	50	0.0008	1.2E-08
Others					
Benzene	8.30E-06	IRIS, 1994	2180	0.035	2.9E-7
1,2-Dibromoethane	2.20E-04	IRIS, 1994	54	0.000864	1.9E-7
Acrylonitrile	6.80E-05	IRIS, 1994	84.3	0.00134	9.2E-08
Cadmium	1.80E-03	IRIS, 1994	0.0086	0.000000137	2.5E-10

(a) Median values as reported in RWDI (1992)

(b) Based on a dilution factor of 1.6E-5; for detail see text

(c) Based on the data reported for 1,2-dichloroethane

(d) Based on data reported for chloromethane

(e) Based on oral RfD

After the series of screenings, the following chemicals remain for further assessment:

Butan-2-ol
Dichlorodifluoromethane
Methylene chloride
Hexane
Toluene
Decane
Butane
Nitrogen oxides
1,1-Dichloroethane
Octane
Chlorodifluoromethane
Pentane
Trichlorofluoromethane
3-Methylheptane
Tetrachloroethylene
Carbonyl sulphide
Methyl mercaptan
Dichlorofluoromethane
MEK (methyl ethyl ketone)
Acetone
Ethyl mercaptan
Cyclohexane
Dimethylsulphide
1,1,1-Trichloroethane
Nonane
MIK (methyl isobutyl ketone)
THF (tetrahydrofuran)
2,3-Dimethylpentane
Carbon disulphide
Butylacetate
Hydrogen sulfide
Dimethyldisulphide
Vinyl chloride
1,1-Dichloroethylene
1,1,2,2-Tetrachloroethane
Bromodichloromethane

Additional Screening for Air Emissions

The number of chemicals remaining for assessment after the original screening is too large to conduct a detailed assessment on each chemical. For this reason, an additional screening tool was used for the landfill parameters.

It is a common practice in human health risk assessment to screen non-carcinogenic chemicals from a detailed assessment by comparing predicted concentrations to reference concentrations. This screening method is also utilized in ecological assessments; however, determining reference concentrations for ecological receptors at a level for screening chemicals is problematic, whereas reference concentrations are more readily available for human health. These reference concentrations are normally derived from concentrations at which no effects on animals were observed in laboratory experiments. This No Observed Effects Concentration (NOEC) is normally divided by a factor (often 10) to account for variation between species, and then by an additional factor (again, often 10) to adjust the number downwards for human populations. The numbers so derived are normally conservative to assure protection of humans. Since safety factors have been used in their derivation to go from laboratory studies on animals to application to people, the use of these numbers as an indicator of toxicity to animals would be generally conservative, and therefore is appropriate for screening for an ecological risk assessment. Therefore, an additional screening, in which chemicals having an exposure ratio (ratio of maximum 24 hr concentration at the receptor to the reference concentration) of less than 0.1 were screened out for further assessment, was conducted. The 0.1 factor adds additional assurance against screening out chemicals that may be borderline and deserving of closer scrutiny. It has been used in ecological risk assessments in the past as the demarcation between the area of no risk and that of potential risk. [Barnthouse and Suter, 1986, and EPA, 1987, referred to in CCME, 1994]. In addition, chemicals to which plant species are highly sensitive, or which are thought to have long-term ecosystem effects not accounted for by such a screening were not excluded. Table 4.8 shows the data used to screen out chemicals on this basis. The data was made available by the Environmental Standards Section of the Standards Development Branch, MOEE.

The chemicals that remain for assessment after use of this screening tool are as follows:

Methyl mercaptan
Ethyl mercaptan
Nitrogen oxides
Vinyl chloride
1,1-Dichloroethylene
1,1-Dichloroethane
1,1,2,2-Tetrachloroethane
Bromodichloromethane

Table D-4.8: Human health-based exposure factors

Non-carcinogenic toxicants						
Chemical	Critical end-point	RfC $\mu g/m^3$	Reference	Vent Concentration (b) $\mu g/m^3$	Max. 24 hr Ground Level Concentration (b) $\mu g/m^3$	Exposure Ratio
Methyl mercaptan	IRR + CNS	1	IWA, 1994	20476	0.328	0.328
Ethyl mercaptan	IRR	1	IWA, 1994	13693	0.219	0.219
Hydrogen sulfide	IRR	1	IRIS, 1994	840	0.0134	0.0134
Butan-2-ol (d)	Hypoactivity & ataxia	300	IRIS, 1994	121000	1.936	0.00645
Nitrogen oxides	Pulmonary function	100	HEAST, 1992	39700	0.635	0.00635
Dimethylsulphide (d)	NA	29	IWA, 1994	8710	0.139	0.00480
Hexane	CNS	200	IRIS, 1994	52700	0.843	0.00420
Pentane	SYS	115	NATICH, 1992	29600	0.474	0.00411
Tetrachloroethylene	Multisite	100	CEPA, 1993	21600	0.346	0.00345
Dichlorodifluoromethane (d)	Reduced BW	600	IRIS, 1994	118000	1.89	0.00314
Toluene	CNS	400	IRIS, 1994	46900	0.750	0.00187
Octane	SYS	350	NATICH, 1992	36100	0.578	0.00165
3-methylheptane	(See octane)	350	IWA, 1994	24900	0.398	0.00113
Butane	Asphyxiant	620	NATICH, 1992	40600	0.650	0.00104
THF	Na (a)	80	MDEP, 1992	4740	0.0758	0.00094
Acetone (d)	Increase liver, kidney W	300	IRIS, 1994	16400	0.262	0.00087
Dimethyldisulphide	Squamous metaplasia	1.5	IWA, 1994	77	0.00123	0.00082
Decane	(see nonane)	1050	IWA, 1994	41500	0.664	0.00063
MIK	Liver & kidney effects	150	HEAST, 1992	5020	0.0803	0.00053

Chemical	Critical end-point	RfC $\mu g/m^3$	Reference	Vent Concentration (b) $\mu g/m^3$	Max. 24 hr Ground Level Concentration (b) $\mu g/m^3$	Exposure Ratio
Dichlorofluoromethane	see diCldiFmethane	600	IRIS, 1994	18500	0.296	0.00049
Trichlorofluoromethane (d)	Survival	900	IRIS, 1994	26000	0.416	0.00046
1,1,1-trichloroethane (d)	Liver	270	HEAST, 1992	6630	0.106	0.00039
2,3-dimethylpentane	see octane	350	IWA, 1994	3290	0.0526	0.00015
Nonane	na	1050	NATICH, 1992	6320	0.101	0.00009
Butylacetate	IRR	700	IWA, 1994	1720	0.0275	0.00003
Chlorodifluoromethane	Kid, adr, pitu	50000	IRIS, 1994	30200	0.483	0.000009
Reproductive, Developmental and Related Toxicants						
*Hydrogen sulfide	IRR	1	IRIS, 1994	840	0.0134	0.0134
*Methylene chloride	Liver toxicity	183	IRIS, 1994	91780	1.47	0.00802
*1,1-dichloroethane (e)	NA	95	CAPCOA, 1993	38912	0.623	0.00650
*Vinyl chloride	NA	26	CAPCOA, 1993	6467	0.103	0.00397
*Bromodichloromethane (d)	Kd cytomegaly	61	IRIS, 1994	13803	0.221	0.00362
*Tetrachloroethylene	Multisite	100	CEPA, 1993	21600	0.346	0.00345
*Dichlorodifluoromethane (d)	Dec BW	600	IRIS, 1994	118000	1.898	0.00314
*Toluene	CNS	400	IRIS, 1994	46900	0.750	0.00187
Carbonyl sulphide	Fetotox/terato	300	IRIS, 1994	20700	0.331	0.00110
*Acetone (d)	Increase liver, kidney W	300	IRIS, 1994	16400	0.262	0.00087
Cyclohexane	NA	280	NATICH, 1992; IWA, 1994	9680	0.155	0.00055
*Trichlorofluoromethane (d)	Survival	900	IRIS, 1994	26000	0.416	0.00046

Chemical	Critical end-point	RfC $\mu g/m^3$	Reference	Vent Concentration (b) $\mu g/m^3$	Max. 24 hr Ground Level Concentration (b) $\mu g/m^3$	Exposure Ratio
*1,1,1-trichloroethane (d)	Liver	270	HEAST, 1992	6630	0.106	0.00039
Carbon disulphide	Fetotox/terato	300	IRIS, 1994	2190	0.0350	0.00011
*Chlorodifluoromethane	Kd, adren, pitu	50000	IRIS, 1994	30200	0.483	0.000009

(a) na: not availble
(b) Mean of values measured in the vent at the Brittania landfill as reported in RWDI, 1992 and IWA, 1994
(c) Based on a dilution factor of 1.6E-5; for detail see text
(d) RfC derived from an oral RfD by assuming 70 kg and 23 m^3/day
(e) Exposure limit is that for 1,2-dichloroethane (derived by CAPCOA based on American Conference of Governmental Industrial Hygenists Threshold Limit values (ACGIH TLV))
*Exposure limit reported is for the most sensitive (critical) end-point
IRR = irritation
CNS = Central nervous sytem
SYS = Systemic toxicant
Dec BW = decreased body weight

Given an emission rate of gases from the landfill, calculations can be made to indicate the potential significance of contributions from the landfill of acid forming gases to Province-wide deposition and acid loadings. The IWA document (Appendix QD, Characterization and Estimation of Chemical Emissions) gives a concentration of NO_x in the flare gases at the Brock West site of between 16 and 21 ppm. Using 21ppm and converting to $\mu g/m^3$ using a factor of 1,724 gives a flare gas concentration of 36,200 $\mu g/m^3$. Modelling by the Research and Technology Branch indicates that total flare gas emissions over at the Durham site 40 years would be $1.39x10^{10}$ m^3. At an NO_x concentration of 36,200 $\mu g/m^3$, this would be a total of 500 tonnes in the 40 years, or 12.5 tonnes/yr. This is 0.0023% of the total emissions in Ontario, and, according to the screening criteria in Section D-4.1.1, would not be considered for further assessment.

The documentation of the IWA does not show any estimates of sulphate emissions for the Durham site, nor are measured concentrations of sulphates reported for other sites. Sulphur is given off from these sites in several reduced forms. There appear to be large differences in the degree to which they are oxidized in the flaring process, varying from 99.9% efficiency for hydrogen sulphide at the Brock West site to only 5% efficiency for carbonyl sulphide at the Beare Rd. site. The efficiency of conversion for methyl mercaptan alone varies from 55% at Beare Rd. to greater than 99.7% at Brock West. Calculations indicate that total concentrations of sulphur contained in compounds would be 38,930 μg S/m^3 in the vent gases. If this were entirely converted to SO_2, it would be 77,600 μg SO_2/m^3. Again assuming total gas production of $1.39x10^{10}$ m^3, this would be a total of 1,078 tonnes in 40 years or 27 tonnes SO_2/yr being emitted from the landfill. This represents 0.003% of total annual emissions in Ontario. Therefore, landfills were not considered as significant contributors to total Ontario loadings.

The following chemicals remain for detailed assessment:

Methyl mercaptan
Ethyl mercaptan
Vinyl chloride
1,1-Dichloroethylene
1,1-Dichlorethane
1,1,2,2-Tetrachloroethane
Bromodichloromethane

Remodelling of Landfill Emissions and Effect on Screening Chemicals

The Science and Technology Branch (STB) conducted detailed modelling of emissions from landfills of the gases that remained after the screening. This information was used to calculate the dispersion factor used to determine the 24-hour ground level concentrations from the vent gas concentrations. The factor determined from the detailed modelling was 0.00012, whereas the factor used in the screening was 0.000016, a 7.5 fold difference. The larger factor was re-applied to the original list of chemicals and those chemicals which would not have been excluded with the new factor determined. From the non-carcinogenic chemicals, only hydrogen sulphide was reinstated to the list as a result of the change; however, from the carcinogenic compounds Methylene chloride, Trichloroethylene, 1,2-Dichloroethane, 1,1,2-Trichloroethane, benzene, and 1,2-Dibromoethane would exceed the risk cutoff of 10^{-6} and were therefore reinstated to the list. The screening is presented here in the manner in which it was conducted rather than with the final modelled dispersion factor as the basis of all calculations. The latter may have provided for a simpler report, but would not present an accurate documentation of the process. (To obtain the final 24hr GLC, one can multiply the 24hr GLC presented in the screening process above by 7.5)

As a result of adjustments made after the detailed modelling, the following is a list of chemicals to be considered for detailed ecological risk assessment for air emissions from the landfill site.

Carcinogens

Vinyl chloride
1,1-Dichloroethylene
1,1-Dichlorethane
1,1,2,2-Tetrachloroethane
Bromodichloromethane
Methylene chloride
Trichloroethylene
1,2-Dichloroethane
1,1,2-Trichloroethane
Benzene
1,2-Dibromoethane

Non-carcinogens

Methyl mercaptan
Ethyl mercaptan
Hydrogen sulphide

D-4.2 Exposure Modelling for Shortlist of Chemicals

D-4.2.1 Exposure Pathways

To determine the potential of the receptors to contaminants, the following exposure routes were assessed;

1) Vegetation - Deposition to soil , uptake to vegetation from soil, deposition on vegetation

2) Vole - ingestion from vegetation, ingestion from soil, inhalation

3) Fox - ingestion from voles, inhalation, water ingestion, soil ingestion

4) Hawk - ingestion from mice, inhalation, water ingestion, soil ingestion

5) Cattle (deer) - ingestion from vegetation, water ingestion, inhalation, soil ingestion

6) a) Earthworm - deposition to soil, uptake to worm,
   b) Woodcock - ingestion from worms.

D-4.2.2 Summary of CAPCOA Model and Results of Modelling

The CAPCOA model was utilized to estimate exposures for the first five scenarios. The assumptions utilized in the CAPCOA model are therefore incorporated into the modelling for this assessment. Since this model does not cover the earthworm-bird system, a model was developed and used for this scenario for mercury and dioxins/furans.

Although the CAPCOA model itself was not altered for this assessment, some of the default parameters were altered to better suit Ontario conditions and the receptors chosen in this study. All the models and the parameters used are presented in Appendix 2. The most notable change was to split up the CAPCOA "intake factor" (IF) into its components of an adsorbtion factor times the animal's lifespan divided by the animals body weight. This allowed the factor to be used for receptors not specifically addressed in the tables in the CAPCOA document.

Summaries of the results from the exposure modelling for the parameters remaining after screening are presented below. For the mercaptans, information necessary to model ingestion was lacking, hence there are no estimates of ingestion doses. The mercaptans are considered to be a problem from an odour perspective at concentrations far lower than those at which other effects are probable; therefore, it is likely that this lack of data and modelling is inconsequential.

References

References for the exposure assessment section are contained in Section D-6.10

D-4.2.2.1 Vegetation

Impact of Incinerator Sources on Vegetation (Grass and Leafy Plants)

Chemical	Ground Level Conc. (24 hr avg.) (ug/m^3)	Soil Concentration (ug/g)	Vegetation Concentration (ug/g)
Benzene	1.3E-01	1.36E-02	3.66E-06
Dioxins/Furans	1.38E-08	9.20E-08	4.61E-09
Mercury	2.15E-02	1.48E-01	2.38E-02
2-Methyl fluorene	7.14E-04	1.16E-03	4.27E-04

Impact of Landfill Sources on Vegetation (Grass and Leafy Plants)

Chemical	Ground Level Conc. (24 hr avg.) (ug/m^3)	Soil Concentration (ug/g)	Vegetation Concentration (ug/g)
Benzene	2.62E-01	2.73E-02	7.36E-06
Bromodichloromethane	1.66	5.37E-01	8.7E-05
1,2-Dibromoethane	6.48E-03	7.11E-02	2.41E-06
1,1-Dichloroethane	4.68	1.52	6.24E-05
1,2-Dichloroethane	5.4E-02	1.75E-02	1.85E-06
1,1-Dichloroethylene	3.57E-01	4.06E-01	2.51E-06
Ethyl mercaptan	1.64	-	-
Methyl mercaptan	2.52	-	-
Methylene chloride	11.0	3.56	9.66E-05
1,1,2,2-Tetrachloroethane	1.66E-01	1.08E-02	1.67E-04
1,1,2-Trichloroethane	6.48E-02	1.44E-01	1.07E-05
Trichloroethylene	1.27	8.3E-02	7.28E-05
Vinyl chloride	7.74E-01	5.06E-02	6.2E-07

D-4.2.2.2 Cattle

Impact of Incinerator Sources on Cattle

Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Veg. Ingestion Dose (ug/kg//d)	Feed Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Cattle (ug/g)
Benzene	1.3E-01		7.84E-05	6.37E-06	1.53E-02	1.54E-02	2.08E-02	1.98E-02
Dioxins/Furans	1.38E-08		9.88E-08	8.03E-09	1.04E-07	2.10E-07	2.21E-09	4.31E-08
Mercury	2.15E-02		5.10E-01	4.14E-02	1.67E-01	7.18E-01	3.44E-03	9.89E-03
2-Methyl fluorene	7.14E-04		9.15E-03	7.43E-04	1.31E-03	1.12E-02	1.14E-04	6.20E-03

Impact of Landfill Sources on Cattle

Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Veg. Ingestion Dose (ug/kg/d)	Feed Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Cattle (ug/g)
Benzene	2.62E-01	5.20E-03	1.58E-04	1.28E-05	3.07E-02	3.61E-02	4.19E-02	4.27E-02
Bromodichloromethane	1.66	6.20E-03	1.86E-03	1.52E-04	6.04E-01	6.12E-01	2.65E-01	4.81E-01
1,2-Dibromoethane	6.48E-03	0	5.17E-05	4.20E-06	7.99E-02	8.00E-02	1.04E-03	4.44E-02
1,1-Dichloroethane	4.68	9.60E-01	1.34E-03	1.09E-04	1.71	2.67	7.49E-01	1.87
1,2-Dichloroethane	5.4E-02	5.60E-03	3.97E-03	3.23E-06	1.97E-02	2.53E-02	8.64E-03	1.86E-02
1,1-Dichloroethylene	3.57E-01	8.40E-03	5.37E-05	4.37E-06	4.56E-01	4.65E-01	5.71E-02	2.86E-01
Ethyl mercaptan	1.64	0	0	0	0	0	2.62E-01	1.44E-01
Methyl mercaptan	2.52	0	0	0	0	0	4.03E-01	2.21E-01
Methylene chloride	11.0	1.00E-02	2.07E-03	1.68E-04	4.0	4.02	1.76	3.16
1,1,2,2-Tetrachloroethane	1.66E-01	3.60E-03	3.57E-03	2.9E-04	1.22E-02	1.96E-02	2.65E-02	2.53E-02
1,1,2-Trichloroethylene	6.48E-02	6.00E-03	2.29E-04	1.86E-05	1.62E-01	1.68E-01	1.04E-02	9.77E-02
Trichloroethylene	1.27	8.60E-05	1.56E-03	1.27E-04	9.33E-02	9.51E-02	2.03E-01	1.63E-01
Vinyl chloride	7.74E-01	3.42E-02	1.33E-05	1.08E-06	5.69E-02	9.11E-02	1.24E-01	1.18E-01

D-4.2.2.3 Field voles

Impact of Incinerator Sources on Field Voles

Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Veg. Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Vole (ug/g)
Benzene	1.3E-01		1.19E-03	4.41E-02	4.53E-02	1.69E-01	1.93E-02
Dioxins/Furans	1.38E-08		1.50E-06	2.99E-07	1.80E-06	1.79E-08	6.04E-08
Mercury	2.15E-02		7.73	4.82E-01	8.21	2.80E-02	1.85E-02
2-Methyl fluorene	7.14E-04		1.39E-01	3.77E-02	1.42E-01	9.28E-04	1.29E-02

Impact of Landfill Sources on Field Voles

Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Veg. Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Vole (ug/g)
Benzene	2.62E-01	5.20E-03	2.39E-03	8.88E-02	9.64E-02	3.40E-01	3.93E-02
Bromodichloromethane	1.66	6.20E-03	2.83E-02	1.74	1.78	2.15	3.54E-01
1,2-Dibromoethane	6.48E-03	0	7.84E-04	2.31E-01	2.32E-01	8.42E-03	2.16E-02
1,1-Dichloroethane	4.68	9.60E-01	2.03E-02	4.93	5.91	6.08	1.08
1,2-Dichloroethane	5.4E-02	5.60E-03	6.02E-04	5.69E-02	6.31E-02	7.02E-02	1.2E-02
1,1-Dichloroethylene	3.57E-01	8.40E-03	8.14E-04	1.32	1.33	4.64E-01	1.61E-01
Ethyl mercaptan	1.64	0	0	0	0	2.13	1.92E-01
Methyl mercaptan	2.52	0	0	0	0	3.28	2.95E-05
Methylene chloride	11.0	1.00E-02	3.14E-02	11.6	11.6	14.3	2.33
1,1,2,2-Tetrachloroethane	1.66E-01	3.60E-03	5.42E-02	3.52E-02	9.29E-02	2.15E-01	2.77E-02
1,1,2-Trichloroethylene	6.48E-02	6.00E-03	3.47E-03	4.67E-01	4.76E-01	8.42E-02	5.05E-02
Trichloroethylene	1.27	8.60E-05	2.37E-02	2.70E-01	2.93E-01	1.65	1.75E-01
Vinyl chloride	7.74E-01	3.42E-02	2.02E-04	1.64E-01	1.99E-01	1.01	1.08E-01

D-4.2.2.4 Red fox

Impact of Incinerator Sources on Red Fox

Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Vole Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Fox (ug/g)
Benzene	1.3E-01		1.35	9.50E-03	1.36	5.20E-02	1.55E-01
Dioxins/Furans	1.38E-08		4.23E-06	6.44E-08	4.30E-06	5.52E-09	4.71E-07
Mercury	2.15E-02		1.30	1.04E-01	1.40	8.60E-03	1.55E-01
2-Methyl fluorene	7.14E-04		9.03E-01	8.12E-04	9.04E-01	2.86E-04	9.91E-02

Impact of Landfill Sources on Red Fox

Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Vole Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Fox (ug/g)
Benzene	2.62E-01	2.18E-03	2.75	1.91E-02	2.77	1.05E-01	1.58
Bromodichloromethane	1.66	2.60E-03	24.8	3.76E-02	25.1	6.62E-01	14.1
1,2-Dibromoethane	6.48E-03	0	1.51	4.97E-02	1.56	2.59E-03	8.58E-01
1,1-Dichloroethane	4.68	4.03E-01	75.6	1.06	77	1.87	43.2
1,2-Dichloroethane	5.4E-02	2.35E-03	8.4E-01	1.23E-02	8.54E-01	2.16E-02	4.8E-01
1,1-Dichloroethylene	3.57E-01	3.53E-03	11.3	2.84E-01	11.6	1.43E-01	6.42
Ethyl mercaptan	1.64	0	0	0	0	6.56E-01	3.59E-01
Methyl mercaptan	2.52	0	0	0	0	1.01	5.52E-01
Methylene chloride	11.0	4.20E-03	163	2.49	166	4.39	93.1
1,1,2,2-Tetrachloroethane	1.66E-01	1.51E-03	1.94	7.57E-03	1.95	6.62E-02	1.11
1,1,2-Trichloroethylene	6.48E-02	2.52E-03	3.53	1.01E-01	3.64	2.59E-02	2.01
Trichloroethylene	1.27	3.61E-05	12.3	5.81E-02	12.3	5.08E-01	7.02
Vinyl chloride	7.74E-01	1.44E-02	7.59	3.54E-02	7.64	3.10E-01	4.36

D-4.2.2.5 Red-tailed Hawk

Impact of Incinerator Sources on Red-tailed Hawk							
Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Vole Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Hawk (ug/g)
Benzene	1.30E-01		1.93	1.36E-02	1.94	1.95E-01	7.02
Dioxins/Furans	1.38E-08		6.04E-06	9.20E-08	6.14E-06	2.07E-08	2.02E-05
Mercury	2.15E-02		1.85	1.48E-01	2.00	3.23E-02	6.68
2-Methyl fluorene	7.14E-04		1.29	1.16E-03	1.29	1.07E-03	4.25

Impact of Landfill Sources on Red-tailed Hawk							
Chemical	24hr Ground Level Conc. (ug/m^3)	Water Ingestion Dose (ug/kg/d)	Vole Ingestion Dose (ug/kg/d)	Soil Ingestion Dose (ug/kg/d)	Total Ingestion Dose (ug/kg/d)	Total Inhalation Dose (ug/kg/d)	Concentration in Hawk (ug/g)
Benzene	2.62E-01	1.52E-03	3.93	2.73E-02	3.96	3.92E-01	11.9
Bromodichloromethane	1.66	1.81E-03	35.4	5.37E-01	35.9	2.48	105
1,2-Dibromoethane	6.48E-03	0	2.16	7.11E-02	2.23	9.72E-03	6.14
1,1-Dichloroethane	4.68	2.80E-01	108	1.52	110	7.02	320
1,2-Dichloroethane	5.4E-02	1.63E-03	1.2	1.75E-02	1.22	8.1E-02	3.56
1,1-Dichloroethylene	3.57E-01	2.45E-03	16.1	4.06E-01	16.5	5.36E-01	46.7
Ethyl mercaptan	1.64	0	0	0	0	2.46	6.73
Methyl mercaptan	2.52	0	0	0	0	3.78	10.3
Methylene chloride	11.0	2.92E-03	233	3.56	236	16.5	692
1,1,2,2-Tetrachloroethane	1.66E-01	1.05E-03	2.77	1.08E-02	2.79	2.48E-01	8.31
1,1,2-Trichloroethylene	6.48E-02	1.75E-03	5.05	1.44E-01	5.19	9.72E-02	14.5
Trichloroethylene	1.27	2.51E-05	17.5	8.3E-02	17.6	1.91	53.3
Vinyl chloride	7.74E-01	9.98E-03	10.8	5.06E-02	10.9	1.16	33

D-4.2.3 Model for maximum exposure to dioxin by American Woodcock

Calculations are based on 1^2 m soil surface

Factor		Units	Value	Calculation
Earthworm uptake of dioxin from soil				
Concentration of dioxin in soil [a]		ng/g	0.000092	A
Amount of soil ingested per year [b]		g	7000	B
Amount of dioxin ingested		ng	0.644	C = A x B
Weight of worms in 1 m^2 [b]		g	150	D
Max conc of dioxin in worms [c]		ng/g	0.0016	E =0.38x C/D
Woodcock uptake of dioxin from earthworms				
Worm ingestion rate [e]		g/day	150	I
No days feeding (seasonal residence)		days	180	J
Concentration in worms		ng/g	0.00163	K
Dioxin load ingested by Woodcock		ng	44.0	L = I x J x K
Weight of bird [d]	Female	g	219	M
	Male	day	0.176	N
Concentration in Woodcock	Female	ng/g	0.20	O = L/M
	Male	ng/g	0.25	P = L/N
Egg dioxin		ng/g	0.00092	
Normal eggs		ng/g	0.0006	
Embryo risk NOAEL		ng/g	0.065	

a = Projected from CAPCOA model
b = Barley, 1961
c = uptake factor of 0.38 calculated from data in Reinecke, 1984
d = Dunning, 1993
e = Terres, 1992

OTR_{98} for 2,3,7,8-T4CDD TEQ	ng/g	0.0048

D-4.2.4 Model for exposure to mercury by American Woodcock

Calculations are based on 1 m^2 soil surface

Factor		Units	Value	Calculation
Calculation of Hg concentration in worms				
Concentration of Hg in soil [a]		ng/g	148	A
Amount of soil ingested per year [b]		g	7000	B
Amount of Hg ingested		ng	1036000	C = A x B
Weight of worms in 1 m2 [b]		g	150	D
Conc of Hg in worms [c]		ng/g	217	E = .0314*C/D
Woodcock uptake of Hg from earthworms				
Worm ingestion rate [e]		g/day	150	I
No days feeding (seasonal residence)		day	180	J
Concentration in worms		ng/g	217	K
Hg load ingested by Woodcock		ng	5859000	L = I x J x K
Weight of bird [d]	Female	g	219	M
	Male	g	176	N
Maximum Concentration in Woodcock	Female	ng/g	26800	O = L/M
	Male	ng/g	33300	P = L/N
Daily Dose	Female	ng/g BW/d	149	Q=E x I/M
	Male	ng/g BW/d	185	R=E x I/N
Estimated Concentration in Woodcock [f]	Female	ng/g	4400	
	Male	ng/g	4500	

Calculation of mercury uptake factor from soil. Data from Helmke, 1979

Worms (#)		15	A
Accumulated fraction of applied Hg/ worm in 46 days		0.000135	B
Fraction in 15 worms		0.002025	C = A x B
Hg added	(ug)	1000	D
Hg in 15 worms	(ug)	2.025	E = C x D
Final weight/worm	(g, wet)	0.8	F
Worm moisture	(%)	71.2	G
Worm weight	(g, dry)	3.456	H = A x F x G
Hg conc. in worm	(ug/g, dry)	0.586	I = E / H
Weight of worms	(g/m2)	150	M
Weight of soil ingested/ year	(g/m2)/yr	7000	N
Weight of soil ingested / weight of worm	(g/g)	46.67	O = N / M
Total worm weight	(g fresh)	12	P = A x F
Soil ingested by 15 worms	(g)	560	Q = O X P
Soil used in test	(g)	1600	R
Proportion of soil ingested		0.35	S = Q / R

Worm activity in field	(days)	250	T
Experiment duration	(days)	46	U
Hg absorption by worms (fraction)		0.03144	E/D x T/U x 1/S

a = Projected from CAPCOA model
b = Barley, 1961
c = includes absorption factor, see calculation below.
d = Dunning, 1993
e = Terres, 1992
f = An iterative approach using a gactrointestinal uptake factor of 0.9 and an Hg rate of loss of 3% of total body burden/day was used to develop this estimate. The 0.9 factor is based on the Hg being in the Methyl form, and is conservative since only a fraction of Hg in worms is Methyl Hg.

D-5. HAZARD ASSESSMENT

D-5.1 Introduction

The screening process demonstrated the potential for the following chemicals to be of concern:

From incinerators	From Landfills	
Benzene	Benzene	Ethyl mercaptan
Dioxins and Furans	Bromodichloromethane	Methyl mercaptan
Mercury	1,2-Dibromoethane	Methylene chloride
2-Methyl-fluorene	1,1-Dichloroethane	1,1,2,2-Tetrachloroethane
NO_x	1,2-Dichloroethane	1,1,2-Trichloroethane
HCl	1,1-Dichloroethylene	Trichloroethylene
	Hydrogen sulphide	Vinyl chloride

To assess the potential effects of the above chemicals on terrestrial systems (at the concentrations and outputs determined in the exposure assessment section), it is necessary to gather existing relevant information. For many of the compounds, very little data exists in the scientific literature on their effects on the natural environment and it is not always possible to provide both a reliable and rapid assessment of their potential hazards. Since this study attempts to assess the potential effects on a large number of terrestrial receptors, this problem is further exacerbated.

D-5.2 Benzene (Incineration and Landfill Source)

D-5.2.1 Vegetation

Data from Currier [1951] have been used to derive an Injury Threshold of 2.89 g/m^3.

Table D-5.1: Calculation of injury to barley after 0.5 hr exposure to benzene after 1 week (computed from data provided in Currier, 1951)

Concentration in Air		
(mol/l)	(g/m^3)	Injury (%)
0	0.0	0
2.2	17.2	30
3.2	25.0	85
6.4	50.0	100

Results of studies with agricultural crops exposed to soil treated with benzene indicate lowest observed effects concentrations (LOECs) for seedling emergence for radish and lettuce (three-day exposure period) to be 274 mg/kg (soil benzene concentration) [Env. Can. 1995]. Root elongation in radish was reduced at soil benzene concentrations of 548 mg/kg over a three-day experiment [Env. Can. 1995].

D-5.2.2 Earthworms

Toxicity of benzene to *Eisenia foetida*

Exposure	Effect Concentration	Endpoint	Reference
Filter paper for 2 days	98 μg/cm2	LC_{50}	Roberts et al., 1984
Filter paper for 2 days	100- 1000 μg/cm2	LC_{50}	Neuhauser et al, 1985
Silt soil in petridish, 6 weeks	4% (w/w) 8% (w/w)	NOEC LOEC	Hartenstein, 1982
Artificial soil, 14 days at 80% moisture	2191 (234) mg/kg 4383 (467) mg/kg 2622 (280) mg/kg 3209 (342) mg/kg	NOEC LOEC LC_{25} LC_{50}	Environment Canada, 1995
Artificial soil, 14 days at 80% moisture	1096 (32)mg/kg 2191 (64)mg/kg 1440 (42)mg/kg 1880 (55)mg/kg	NOEC LOEC* LC_{25} LC_{50}	Environment Canada, 1995

( ) value in brackets is concentration measured at the end of the experiment

* 60% mortality

D-5.2.3 Cattle

No information available

D-5.2.4 Field Voles

Slooff [1988] reported that the acute toxic concnetration of benzene to rats is 41.6 g/m³. The LD_{50} for rats fed benzene in their diet over a 96 hour period was 3300 mg/kg bw (body weight) [Mallinckrodt, 1989]. The lethal dose (LC_{50}) for rats was determined to be 5600 mg/kg b.w. [Jorgensen]

Table D-5.2: Summary of effects of benzene on mammals [Extracted from IRIS database]

Species	Exposure	Response	Reference
Rat	0, 50 ,250, 500 mg/kg b.w.	Increase in mammary tumours & Zymbal gland and oral cavity carinomas, and leukemia	Maltoni et al., 1979 Maltoni et al., 1983
	0-200 mg/kg b.w. 5x/wk for 103 wks	Increases in Zymbal gland carcinoma, oral cavity carinoma, and skin tumors (males)	NTP, 1986
	300 ppm benzene 6 hr/day,5 days/wk for 488 days (inhalation)	No increase in neoplasia incidence	Snyder et al., 1981
	300 ppm 7 hrs/day for 12 weeks (inhalation)	Increased incidence of hepatomas and Zymbal gland carcinoma,	Maltoni, 1983
Mouse	0-200 mgkg b.w. 5x/wk for 103 wks	Increases in Zymbal gland carcinoma, lymphomas, & lung tumors, Males had harderian & preputial gland tumors, females had mammary & ovary tumors.	NTP, 1986
	300 ppm benzene 6 hr/day,5 days/wk for 488 days (inhalation)	Slight increase in hematopoietic neoplasms for some strains of mice.	Snyder et al., 1981

D-5.2.5 Red Fox

No information available

D-5.2.6 Red-tailed Hawk

No information available

D-5.3 Dioxin/Furans (Incineration Source)

D-5.3.1 Vegetation

Several studies have shown that TCDD is not readily taken up and translocated by terrestrial plants [Kenaga, 1983]. Tsushimoto [1981], cited in Kenaga [1983] found no toxicity to the pondweeds *Elodea nuttali* and *Ceratophyllum emersum* containing 7000 ppt TCDD.

D-5.3.2 Earthworms

Earthworms at the dioxin spill site at Seveso, Italy were found to bioaccumulate dioxin with a linear relationship between the concentrations in soil and worms [Martinucci, 1983]. Bioaccumulation factors ranged from 5 to 42 The dioxin concentrations in worms ranged from non-detectable at control sites to a maximum of 248 μg/kg. The earthworm populations did not appear to be affected but their activity may have been slightly depressed in the most heavily impacted areas where soil concentrations were as high as 9.3 μg/kg TCDD. To some extent, the earthworms in the impacted zone may have benefited from reduced numbers of predatory species due to dioxin toxicity. Analysis of the species composition suggested that *Allobophora rosea* may be more sesitive to dioxin than *A. chlorotica.*

Soil concentrations of 5 μg/g dioxin for 85 days or tissue levels of >2200 μg/g were found to cause mortality in earthworms [Reineke & Nash, 1984].

D-5.3.3 Cattle

Studies by Furst et al. [1993] suggest that an exposure pathway for dioxins through an air-grass-cow route may be more important than for the soil-grass-cow pathway. Soil concentrations up to 30 pg l-TEq/g did not cause noticably elevated PCDD/PCDF levels in the cow's milk. The T,CDD carry over factor of grass to milk was approximately 4 whereas the carry over factor for O,CDD was 0.1. They recommended that risk assessment should be based on a congener specific basis rather than on toxic equivalents because accumulation is highly dependent on congeners present.

Holstein cattle exposed to radio-labelled 2,3,7,8-Tetrachlorodibenzo-p-dioxin either in soil or grain at 0.05 μg/kg body weight did not show any loss of weight within 14 days of treatment [Jones et al., 1989]. Hepatic AHH and EROD activity was slightly greater than those for the control; however, caution should be execised in interpreting the data because of the small number of samples. The dioxin residues in tissues and milk were generally similar regardless of source (soil or grain). By the end of the test (at 14 days), the percent of administered dioxin was similar for both treatments at just over 13%. The highest concentrations of dioxin accumulated in the fat with a maximum of approximately 0.15 μg/kg. The muscle, liver, kidney, and brain tissues all generally contained below 0.01 μg/kg TCDD.

D-5.3.4 Field Voles

Table D-5.3: Mamalian toxicity of dioxin reported in the literature

Species	LD_{50} oral (μg/kg)	LD_{50} Tissue (μg/kg)	Whole body half life (days)	References
Guinea pig	0.6 -2.1			Poiger et al., 1982.
	>5<10		>20	Decad et al., 1982.
Guinea pig		1-2		Smuckler, 1985.

Species	LD_{50} oral (μg/kg)	LD_{50}Tissue (μg/kg)	Whole body half life (days)	References
Rat	22-45			Poiger et al., 1982.
			<2	Decad et al., 1982.
Hamster		2000-5000		Smuckler, 1985.
Mouse	280			Poiger et al., 1982.
Rhesus monkey	1000		8	Decad et al., 1982.

- Acute oral toxicity of TCDD is between 0.6 and 0.8 μg/kg b.w. for guinea pigs
- 1% of TCDD body burden is excreted per day by rats.

D-5.3.5 Red Fox

No information available

D-5.3.6 Red-tailed Hawk

Table D-5.4: Avian toxicity of TCDD

Species	Dose	Response	Age	Reference
Chicken, White Leghorn	25-50 μg/g	LD_{50}	Adult	Greig et al., 1973
	18 μg/g (oral)	Hydropericardium		Schwetz et al., 1973
	10 μg/g (i.p.)	Involution of Bursa of Fabricius		Sawyer et al., 1986
	1.8 μg/g (oral)	NOEL		Schwetz et al., 1973
	0.3 μg/g (i.p.)	AHH induction (ED_{50})		Sawyer et al., 1986
	0.04- 0.14 μg/kg (total in egg)	LD_{50}	Embryo	Cheung et al., 1981
	0.065 μg/kg (total in egg)	Cardiovascular malformations	Embryo	Cheung et al., 1981
	0.06- 0.09 μg/kg (total in egg)	AHH induction (ED_{50})	Embryo	Poland et al. 1973, 1976; Rifkind et al., 1985
Eastern Bluebird	10 μg/kg	100% hatching failure	Egg	Martin et al., 1989
Ring-necked Pheasant	<25 μg/kg (i.p.)	LD_{50} single dose	Adult	Nosek et al., 1992
	<10 μg/kg (i.p.)	LD_{50} cumulative dose	Adult	Nosek et al., 1992
	4.5 μg/kg (i.p.)	50% reduction in hatchability	Egg	Nosek et al., 1992
	1 μg/kg (i.p.)	NOEL cumulative dose	Adult	Nosek et al., 1992
American Robin	0.3-1 μg/kg (i.p.)	AHH/EROD induction (doubling or more of control levels)	Adult	Martin et al., 1989
Bobwhite	15 μg/g (single oral)	LD_{50}		Eisler, 1986
Mallard	> 108 μg/kg (oral, acute)	LD_{50}		Eisler, 1986
Ringed Dove	> 810 μg/g (single oral)	LD_{50}		Eisler, 1986
Great Blue Heron	0.14-0.19 μg/g -egg	LOEL	Embryo	Hart, et al., 1991

TCDD residue concentrations in bird tissues at a contaminated study site at Elgin, Florida [Young and Cockerham, 1985]

Bird Species	Liver (ppt)	Stomach (ppt)
Southern Meadowlark	440	10
Mourning Dove	50	10
Savannah Sparrow	69	84

Chickens exposed to dioxin and furan-contaminated soil showed no adverse effects for up to 278 days [Petreas et al., 1991]. The exposure consisted of contaminated soil mixed with a formulated diet amended at a rate of 10% soil. The congener compositions varied among treatments (control/background, high and intermediate); however, the concentrations in the prepared diets ranged from 1.1 ppt 2,3,7,8-TCDD to 9518 ppt OCDD and 8.8 ppt 2,3,7,8 TCDF to 704.8 ppt OCDF in the high exposure treatment.

The data showed dioxins in the eggs with values of 8.2 pg/g fat in the case of 2,3,7,8 TCDD and 17570 pg/g fat for OCDD. The ratios between concentrations in eggs and original soil content ranged from 0.77 to 0.14 and showed diminishing values with increasing degrees of chlorination. This study demonstrated that chickens can accumulate dioxins and furans at very low levels in soil and transfer them to the eggs in proportion to the exposure. On the otherhand, there appears to be some normal background level for dioxins in chickens that is derived from ambient soil or food, even in uncontaminated areas.

D-5.4 Mercury (Incineration Source)

D-5.4.1 Vegetation

The proposed soil cleanup guideline for contaminated sites has been set at 10 μg/g based on the protection of vegetation. This criterion is supported by studies in the literature which indicate that foliar injury and/or growth effects occur at levels above 10 μg/g. Studies are listed in the Appendices which summarize mercury effects on terrestrial plants [Weaver et al. 1984, Sheppard et al. 1993].

D-5.4.2 Earthworms

Literature data concerning mercury and earthworms indicates that worms accumulate mercury from the soil, however, the data are variable. Perhaps the variability is related to species differences as the literature references each involved a different worm species. In order to compare data presented in the reports, the concentrations were computed using a factor of 71.2 % moisture based on work conducted by MOEE [MOEE data].

Beyer et al. [1985] indicated that soil methyl mercury concentrations above 25 ppm were lethal to *Eisenia foenda* earthworms while concentrations of 5.0 ppm decreased the ability of worms to regenerate new segments if injured. At 1 ppm Hg, there was no statistical decrease in regeneration ability. Assuming that this is the critical concentration for mercury, concentrations of mercury in tissues above 150 μg/g may indicate toxicity. The bioconcentration factor for mercury from soil was 21.6 (wet weight basis) or 93 (dry weight basis).

Bull et al. [1977] reported that *Lumbricus terrestris* collected in the vicinity of a chlor-alkalai plant accumulated higher concentrations of mercury that at control sites. On a dry weight basis, the BCF was 1.49 to 1.7 from soil to worm. On a wet weight basis, the BCF was 0.32 to 0.40. There was no indication that toxicity occurred at the highest concentration (1.29 ppm fresh weight, 5.6 dry weight) found in the worms nearest the mercury source.

Studies of mercury uptake by *Aporrectria tuberculata* in soils treated with sewage sludge containing mercury (6.8 ppm) and other metals showed increasing levels of mercury in the worms and castings with increasing rates of sludge applied [Helmke, 1979]. The maximum concentration in the worms was 0.76 ppm and there was no indication that these levels were toxic. The concentrations in the worms were higher than in the casting and soil. The concentrations of mercury in the soil (computed from loadings of sludge) was lower than those measured in the worm casts. The reason for this is uncertain, especially if, as expected, the worms were removing mercury from the soil as it passed through their bodies. *L. terrestris* had 8-13% of the mercury present in the methyl form.

In one study, the LC_{50} for earthworms (*Octochaetus patton*i) exposed to Hg in soil for 60 days was 0.79 mg/kg d.w. (dry weight) while the LC_{100} was 5 mg/kg[Abbasi and Soni, 1983]. In earthworm study, the NOEL was 96 mg/kg d.w. over a 14 day period, whereas 58% mortality occurred in earthworms exposed to 194 mg/kg d.w. in the same study [Env.Can. 1995].

D-5.4.3 Cattle

Data pertaining to the effects of mercury on cattle were not available; however, studies with pigs indicate that a NOEC was observed at a single acute oral dose of Methylmercuric dycyandiamide (MMD) at 1.7 mg Hg/kg feed [Piper et al. 1971]. At oral doses of 3.4 mg Hg/kg, pigs exhibited CNS depression, loss of weight, weakness, vomiting staggering and stiffness, but were able to recover. Piper et al. [1971] suggested that the minimum acute lethal oral dose is 13.4 mg Hg/kg. Typhonas and Nielsen [1973] also looked at the effects of mercury on pigs and determined a LOEL at a dose of 0.38 mg Hg/kg over a 60 to 90 day exposure.

D-5.4.4 Field Voles

For laboratory rats exposed to Hg in their diet over a 12 week period, the LOEL was 20 mg/kg b.w. for unspecified toxicity symptoms and weight loss [Verschueren, 1976]. Rats exposed to 0.4 mg/kg bw Hg over a longer exposure period (2 years) exhibited symptoms of growth reduction, kidney damage and histochemical changes [Verschueren, 1976]. Mitsuromi et al. [1990] found that mice exposed to 2 mg Hg/kg feed over 2 years exhibited chronic kidney damage.

D-5.4.5 Red Fox

No literature pertaining to the Red Fox was found, however, a study was carried out using cats, a predator similar in size to the fox. In this study, the LOEL for cats fed Hg orally over a two year period occurred at tissue concentrations of 46 μg/kg bw [Charbonneau et al. 1976].

D-5.4.6 Birds (e.g. Red-tailed Hawk and Woodcock)

The exposure of birds to mercury is typically through ingestion, drinking water, food and inhalation from air. By far, the greatest effort has been to relate biological effects of mercury ingestion from food to accumulation of the metals in body tissues, physiological and morphological changes and reproduction. Within this body of knowledge, special emphasis has been placed on methyl mercury because this is generally accepted as the most toxic form and because it is commonly encountered as the form being ingested (ie. on seed grain treated with methyl mercury and the occurrence in body tissues. The following text is a summary of the information that may be considered in assessing impacts and risks of mercury contamination of the environment.

Diet

The majority of controlled studies involved feeding various bird species with a diets containing mercury in several forms. The findings of these studies are summarized in the following table.

Table D-5.5: Response of birds to diets containing mercury (mainly cited in Eisler, 1987)

Species	Dietary Exposure	Effect	Reference
American Black Duck	Dietary mercury at 3 mg/kg as methyl mercury over 2 years	Egg production reduced by 35.3% Reproduction inhibited Development of brain lesions Ducklings surviving to 4 weeks reduced by 84.4%	Finley and Stendell , 1978
European Starling	Diet with 1.1 mg/g in food after 8 weeks	Development of kidney lesions	Osborn, 1984
Grey Pheasant	Organomercury 0.640 mg/kg b.w./day in diet for 9 weeks	Reproduction inhibited after feeding for 30 days	McEwen et al, 1973
Japanese Quail	Inorganic mercury with 1.0 or 3.0 mg/kg in food for 9 weeks	Affected brain and plasma enzymes	Hill and Soares, 1984 cited in Eisler, 1987

Species	Dietary Exposure	Effect	Reference
	Inorganic mercury with 8.0 mg/kg in food for 3 weeks	Depressed gonad weight	Hill and Soares, 1984 cited in Eisler, 1987
Ring-necked Pheasant	Methyl mercury in food at 16 mg/kg	ED_{50}	Fimreite, 1971
Rock Dove	Methyl mercury in food at 1 mg/kg body weight/day (= 12 mg/kg in food) for 5 weeks	Behavioral changes	Evans et al 1982 cited in Eisler, 1978
	Inorganic mercury at 3.0 mg/kg body weigh in diet for 17 days	Behavioral changes	Leander et al., 1977 cited in Eisler, 1987

Drinking Water

The only report specifically identifying drinking water as the source of mercury involved chickens. A concentration of 500 mg/l in the water caused decreased growth and affected blood chemistry within 3 days [Grissom and Thaxton, 1985 cited in Eisler, 1987].

Uptake into Tissues

Liver

The concentrations of mercury in liver tissues is a useful measure of exposure, and mercury generally occurs at the highest concentrations in the internal organs. A summary of data relating to mercury content of bird livers is presented below. The data are grouped into normal or background values, mercury concentrations found under contaminated situations, and controlled experimental relationships and effects.

Table D-5.6: Reported background concentrations of mercury in livers of birds

Species	Situation	Concentration	Comment	Reference
American Black Duck	Normal	0.588 mg/kg FW 0.478 mg/kg FW	Male Female	Gochfield & Burger, 1987
American Avocet	Control site Dredge-pit site	0.10 mg/kg FW 0.10 mg/kg FW	Same values at dredge site and control site	White & Cromartie, 1985

Species	Situation	Concentration	Comment	Reference
American Coot	Pre-operational		Minor increase post start-up near power plant and control	White et al., 1986
	Ash pond	0.02 mg/kg FW		
	Control	0.03 mg/kg FW		
	Post-operational			
	Ash pond	0.05 mg/kg FW		
	Control	0.08 mg/kg FW		
Black-necked Stilt	Dredge-pit site Control site	0.10 mg/kg FW 0.10 mg/kg FW	Same values at dredge site and control site	White & Cromartie, 1985
Canvasback	Wild	0.400 mg/kg FW	Normal	Fleming, 1981
Mallard	Normal	0.307 mg/kg FW 0.347 mg/kg FW	Female Male	Gochfield & Burger, 1987
Pintail	Wild	0.055-9.51 mg/kg FW	Normal	Krapu at al.., 1973
Scaup	Normal	0.815 mg/kg FW 0.498 mg/kg FW	Female Male	Gochfield & Burger, 1987
Teal, Green-winged	Control site	0.01 mg/kg FW	Normal	White & Cromartie, 1985

Contaminated sites

The use of mercury-based seed dressings to protect plant seedlings against fungal disease was formerly a common practice. Many wild birds consumed the funicide-treated seed in the field and thus ingested mercury. Fimreite [1970] determined that many species did accumulate mercury in their tissues. An average concentration of mercury in the liver for several species was 1.16 mg/kg FW. Birds of prey, which consumed the seed-eating birds for food contained, on average, almost twice as much mercury (2.30 mg/kg FW) in their livers, indicating that mercury was bioaccumulating in the food chain.

The environmental situation at the English River-Wabigoon Lake area has received a great deal of attention since the discovery of significant releases of mercury from a local pulp and paper mill. The contamination resulted in elevated mercury levels in fish and other biota. Studies by Fimreite [1974] determined that birds (10 species) in the area had an average of 22.50 mg/kg total mercury in their livers. The highest value (96.7 mg/kg FW) for any species was obtained for Turkey Vulture which is a carrion scavenger. This is significant in that these vultures would feed on any animals that might have died as the result of mercury poisoning however there was no indication that the sampled birds showed any toxicological symptoms.

The study also showed that a significant percentage of the mercury was methylated (Mallard, 42.0 %; Pintail, 50.0%, Common Goldeneye, 24.8%; Common Merganser, 15.4%) [Fimreite, 1974]. This pattern is in general agreement with a study of birds in a salt marsh six years after a local chlor-alkali industrial plant closed [Gardner et al, 1978]. The liver mercury concentrations for 12 bird species ranged from 2.4 to 37 mg/kg FW with an average of nearly 60% occurring as methyl mercury.

The maximum mercury concentrations in livers of Black-necked Grebes wintering in a mercury-polluted lagoon was 57.9 mg/kg DW [Leonzio et al, 1986]. By contrast, livers of Black-necked Stilts at a dredge-pit site and a control site were both 0.10 mg/kg FW [White and Cromartie, 1985]. The livers of Herring Gulls foraging at inland landfill sites contained intermediate levels of mercury (7.47-8.38 mg/kg DW) [Leonzio et al., 1986]. Gulls foraging at marine sites had slightly higher levels (13.08-13.30 mg/kg DW), presumably because their diet contained more fish.

Experimental Exposures

Several good experimental studies have been conducted to determine the effects of mercury ingestion by birds. Mercury levels in the livers of American Black Ducks fed 3 mg/kg methyl mercury in their diet for 2 years developed elevated mercury levels in their liver (23.07 and 21.33 mg/kg FW for male and female respectively) [Finley and Stendell, 1978]. These levels were associated with impaired reproduction and brain lesions. European Starlings fed a diet containing 1.1 mg/kg mercury accumulated 6.50 mg/kg in the liver after 8 weeks and developed kidney lesions [Eisler, 1987].

In a study of Red-tailed Hawks, the birds were subjected to a diet of chicks containing 3.9 to 10 mg/kg methyl mercury in their livers [Fimreite and Karstad, 1971]. Hawks demonstrated nerve damage when their own livers reached 3.9 mg/kg mercury. A few hawks died during the study. The lowest mercury concentration associated with the death of these hawks was 16.7 mg/kg in the liver.

Based on work with Ring-necked Pheasant [Fimreite, 1971], a relationship between mercury levels in the liver and food consumed was developed.

Concentration of mercury in liver = $0.071 + (0.343 \times X)$ mg/kg mercury

where X = mercury concentration in food

Another useful relationship found for Alberta/Saskatchewan birds of prey is liver:egg mercury concentration ratio of 4.47. [Fimreite, 1970]. Vermeer [1971] found a slightly higher ratio (5.5) in the case of California Gulls.

Brain

Finley and Stendell [1978] fed American Black Duck on a diet containing 3 ppm methyl mercury for 2 years. This resulted in mercury concentrations in the brain tissue of 2.83 mg/kg FW for males and 3.79 mg/kg FW in females. Control values for both sexes were below detection limit of 0.05 mg/kg FW. By comparison, values as high as 14.91 mg/kg DW were found in brain tissues of Black-necked Grebe wintering in a mercury polluted lagoon. The higher values typically occurred in April reflecting winter feeding at that site [Leonzio et al. 1986].

Leonzio et al [1986] also compared Herring Gulls at marine areas and at inland landfill sites. The concentrations of mercury in brain tissues was 2.74 to 3.03 mg/kg DW in the inland population compared to 3.07 to 3.95 mg/kg DW at the marine locations. The differences were explained on the basis of the presumed higher proportion of fish in the diet (hence higher mercury exposure) at the marine sites

Mercury concentrations in the brain tissue of American Black ducklings was 0.98 mg/kg FW however, 84.4% of the ducklings did not survive to 4 weeks [Finley and Stendell, 1978].

Mercury toxicity is largely related to neurological effects. These include weakness of the extremities and loss of muscle coordination. Microscopic examination of brain tissues of Red-tailed Hawks exposed to mercury showed swelling of myelinated nerves in the spinal chord and assorted other lesions and damage to the brain and associated tissues [Fimreite and Karstad, 1971]. Unfortunately, no analytical tests could be performed to establish brain levels of mercury.

Fat

Data in the literature regarding the deposition of mercury in fatty tissues is limited but can be justified on the basis that mercury concentrations in fat are quite low in relation to other bird tissues. The concentrations of mercury in fat in Black-necked Grebes wintering in a mercury-polluted lagoon ranged seasonally from 0.53 to 1.50 mg/kg DW with the maximum concentrations occurring in April [Leonzio et al., 1986]. There was little difference between foraging sites either at landfills or marine sites with respect to mercury concentrations in the fat of Herring Gulls (range of 0.59 to 1.19 mg/kg DW [Leonzio et al., 1986]. The equivalent concentrations for Black-headed Gulls using the same sites was lower at 0.30 to 0.33 mg/kg DW.

Kidney

American Black Ducks fed a diet with 3 mg/kg methyl mercury for 2 years accumulated kidney levels of 11.16 mg/kg FW (male) and 16.05 mg/kg FW (female) [Finley and Stendell, 1978]. The kidneys of Black-necked Grebes wintering in a mercury-polluted lagoon accumulated up to 35 mg/kg DW mercury over winter in comparison to a low concentration of 7.63 mg/kg DW [Leonzio et al., 1986]. After 8 weeks of exposure to a diet containing 1.1 mg/kg of mercury, European Starling kidneys accumulated 36.3 mg/kg DW mercury and kidney lesions developed [Nicholson and Osborn, 1984 quoted by Eisler, 1987]. The kidney of Herring Gulls foraging at inland landfills accumulated 6.73 to 7.47 mg/kg DW mercury while those at marine locations reached 10.65 to 10.81 mg/kg DW presumably because of higher mercury levels in their diet with a higher content of fish [Leonzio et al. 1986].

Muscle

Male American Black Ducks fed 3 mg/kg methyl mercury in diet for 2 years developed 4.03 mg/kg FW mercury in the muscle tissue [Finley and Stendell, 1978]. The equivalent value for females was 4.51 mg/kg FW. In the same study, duckling muscle contained 0.79 mg/kg FW mercury. Under the experimental conditions, duckling survival at 4 weeks was reduced by 84.4% compared to the control. In view of this, ducklings are less tolerant of mercury than their parents.

Six years after closure of a mercury source, twelve bird species were examined in a nearby salt marsh [Gardner et al. 1978]. The mercury content of muscle tissues in the birds ranged from 0.43-7.4 mg/kg DW with an average of nearly 80% as the methyl form. Mercury in the muscle of Black-necked Grebes in a mercury-polluted lagoon ranged from 3.7 to 11.83 mg/kg DW [Leonzio et al., 1986]. The highest values were measured in April indicating that the birds accumulated mercury over the winter.

Herring Gulls foraging at marine sites accumulated 4.00 to 4.95 mg/kg DW mercury in the muscle whereas birds at inland landfills reached concentrations of 2.57 to 3.54 mg/kg DW [Leonzio et al., 1986].

The lowest mercury concentration in hawk muscle associated with death of Red-tailed Hawks fed on chicks containing methyl mercury in their liver was 4.3 mg/kg mercury [Fimreite and Karstad, 1971]. Hawks with lower levels of mercury in their muscle appeared normal.

Eggs

Maternal mercury as it is deposited in bird eggs. While this does have some benefit to the female laying the egg in that a portion of the body burden of mercury is removed, the embryo is immediately at a disadvantage with its own body burden of mercury. At higher levels, the mercury causes damage to the embryo and can result in death of the chick or failure of the egg to hatch. When this happens, there is a loss of part of the bird population and the considerable investment of resources and time of the parents, particularly the female, is lost.

One aspect of mercury contamination is the reduced size of eggs when adult birds are exposed to mercury. Fimreite [1971] determined that egg weight of Ring-necked Pheasants was reduced by mercury according to the following formula

$$\text{Egg Wt (g)} = 32.88 - (0.285 * X)$$

were X = the concentration of mercury in the food in mg/kg (FW)

Additional information involving eggs and mercury is presented under the section of this document that deals with excretion of mercury.

Feathers

Hg is deposited in feathers during their formation therby serving as an indicator exposure to Hg [Goede, 1984]. The deposition of mercury in the feathers can be interpreted as beneficial as this serves as a means of eliminating mercury from the bird's body (see below).

Table D-5.7: Reported concentrations of mercury in bird feathers

Species	Hg Concentration mg/kg FW	Response	Reference
American Black Duck*	Male 65.70 Female 56.12 Duckling 24.80	Severe reproductive problems	Finley and Stendell, 1978
Bird sp.	40	Impaired reproduction	Spann et al. 1972 (cited in Eisler, 1987)
Bird sp.	5.00-1.10	Reduced hatching and sterility	NAS 1978 (cited in Eisler, 1987)
Sparrow Hawk (*Accipiter nisus*)	40	Sterility	Solonen and Lodnius, 1984 (cited in Eisler, 1987)
*Fed 3 ppm methyl mercury in their diet for 2 years			

Excretion

In chickens, the rate of Hg excretion was reported to be about 3% of the total body burden/day [Ulfvarson, 1962].

Male Herring Gulls are able to excrete 49% of their Hg intake during feather moult [Lewis et al., 1993]. By comparison, excretion by female Herring Gulls is 58% of the Hg intake, the differential being the amount lost during egg laying [Lewis et al., 1993]. The benefit to the female laying the egg is offset by the additional hazard imposed on the developing embryo.

Larger decreases in body burden of mercury have also been estimated for other bird species during molt:

Table D-5.8: Body burden loss of mercury by birds during moult [after Lewis et al., 1993]

Species	Loss	Citation
Black-eared Kite	70%	[Honda et al. 1986]
Black-headed Gull	65%	[Lewis and Furness, 1991]
Bonaparte's Gull	93%	[Braune and Gaskin, 1987]
Herring Gull	58%	(Lewis et al., 1993

Values for use in computing excretion rates of mercury from medium sized birds of prey

The proportion of total body weight that was made up of feathers in a female Bald Eagle was 14.3% [Broadkorb, 1955]. This is higher than the 4-12% range for birds quoted by Terres [1991]. Lewis et al [1993] reported that Herring Gulls had about 10% of their body weight as feathers.

Specific Gravity for eggs of Northern Harrier is 1.038 g/ml

Water content of fresh eggs of Lesser Black-backed Gull is 73% but this decreases to 56% before hatching. This loss of water is equivalent to a 23% increase in concentration of mercury, assuming that no mercury is volatilized from the egg.

Based on the dimensions (59 x 47 mm) given by Harrison [1978] and the equation of [Bergtold, 1929], the weight of a typical Red-tailed Hawk egg is 71 g.

The average clutch size for Red-tailed Hawk in Ontario is 2.4 eggs [Peck and James, 1983].

Biological Effects

Lethal Dose

The toxic doses of mercury for birds are presented in Table D-5.9 below.

Reproduction (survival, hatching)

On a 3-week diet containing 8 mg/kg inorganic mercury, Japanese Quail showed depressed gonad weight [Hill and Soares, 1984 quoted in Eisler, 1987]. Reproduction in Grey Pheasant was inhibited after feeding for 30 days on a diet containing organomercury at a rate 0.6400 mg/kg body weight [McEwen et al, 1973].

In a study of American Black Ducks fed a diet containing 3 mg/kg as methyl mercury over 2 years, egg production was reduced by 35.3%. From this same experiment, the 84.4% reduction of ducklings surviving to 4 weeks was associated with 0.98 mg/kg FW in the brain, 24.8 mg/kg FW in feathers, 4.23 mg/kg in liver and 0.79 mg/kg FW in the muscle tissue [Finley and Stendell, 1978]. The mercury level in the eggs which produced these results averaged 5.10 mg/kg FW.

Mallard ducks exposed to mercury levels of 0.50 mg/kg in their diet for three generations suffered negative reproductive behaviour [Heinz, 1979 quoted in Eisler, 1987]. In this case, the mercury residues in the egg were 0.780-0.860 mg/kg FW, and hatching was reduced.

Based on data for Ring-necked Pheasants experimentally fed methyl mercury [Fimreite, 1971] the following equation was developed:

$$\text{No. young surviving} = 0.954 - (0.028 * X)\ \text{mg/kg Hg}$$

where X is the dietary level of Hg

Eggs of Ring-necked Pheasant with mercury residues of 0.9 to 3.1 mg/kg FW had reduced hatching rates [Spann et al. 1972 quoted in Eisler, 1987]. Mercury residues of 0.79 mg/kg in eggs were related to impaired reproduction by bird species [Spann et al. 1972 quoted in Eisler, 1987]. Mercury residues of 1.3 -2.0 mg/kg FW in eggs of Common Loon caused reduced hatching [Fimreite, 1979 quoted by Eisler, 1987]

An investigation of mercury released from a pulp and paper mill in Northwestern Ontario showed that aquatic bird species were affected by the contaminant. Of note here is the reproduction success of Common Terns in relation to the mercury concentration in their eggs.

Reproduction of Common Terns in Northwestern Ontario in relation to mercury content of eggs [After Fimreite, 1974]

Reproduction Success	Mercury in Eggs (mg/kg)	
	Total Hg	Methyl Hg
Normal fledging	1.00	0.82
Fledging reduced by 90%	3.65	2.4

Mercury residues of 40.0 to as low as 5.0 mg/kg in bird feathers have been associated with impaired reproduction [Spann et al. 1972 and others] while concentrations from 1.10 to 5.00 mg/kg Hg reduced hatching and sterility [NAS 1978 quoted in Eisler, 1987]. Sterility of the Sparrow Hawk (*Accipiter nisus*) was reported when mercury residues in feathers reached 40 mg/kg FW [Solonen and Lodnius, 1984 quoted in Eisler, 1987].

Table D-5.9: Summary of mercury toxicity data reported in the literature (mainly from Eisler, 1987)

Species	Exposure	Hg form	mg/kg b.w.	Effect	Reference
Chicken	Acute Oral	Phenyl	60.0	LD$_{50}$ at 14 days	Mullins et al, 1977
Chukar	Acute Oral	Ethyl	26.9	LD$_{50}$ at 14 days	Hudson et al., 1984
Coturnix	Acute Oral	Methyl	11.0	LD$_{50}$ at 14 days	Hill, 1981
	Acute Oral	Inorganic	26.0	LD$_{50}$ at 14 days	Hill, 1981
	Dietary at 9 wks	Methyl	4.0	LD0	Hill, 1981
	Dietary at 5 days	Methyl	8.0	Some deaths	Hill et al., 1981
	Dietary at 5 days	Methyl	31.0	LD$_{50}$	Hill, 1981
	Dietary at 5 days	Inorganic	2956	LD$_{50}$	Hill, 1981
	Dietary at 9 wks	Inorganic	32.0	LD$_0$	Hill, 1981
Fulvous Whistling Duck	Acute Oral	Methyl	37.8	LD$_{50}$ at 14 days	Hudson et al., 1984
Gray Partridge	Acute Oral	Ethyl	17.6	LD$_{50}$ at 14 days	Hudson et al., 1984
House Sparrow	Acute Oral	Methyl	12.6	LD$_{50}$ at 14 days	Hudson et al., 1984
Japanese Quail	Acute Oral	Methyl	14.4	LD$_{50}$ at 14 days	Hill et al., 1981
	Acute Oral	Ethyl	21.4	LD$_{50}$ at 14 days	Hudson et al., 1984
	Acute Oral	Inorganic	31.1	LD$_{50}$ at 14 days	Hill et al.,1981
	Dietary at 28 day	Inorganic	500	LD$_{33}$	El-Bergearmi, 1980
	Dietary at 28 day	Inorganic	500	LD$_{86}$	El-Bergearmi, 1980
Mallard	Acute Oral	Methyl	2.2	LD$_{50}$ at14 days	Hudson et al., 1984
	Acute Oral	Ethyl	75.7	LD$_{50}$ at 14 days	Hudson et al., 1984
	Acute Oral	Phenyl	524.7	LD$_{50}$ at 14 days	Hudson et al., 1984
Northern Bobwhite	Acute Oral	Methyl	23.8	LD$_{50}$ at 14 days	Hudson et al., 1984
Prairie Chicken	Acute Oral	Ethyl	11.5	LD$_{50}$ at 14 days	Hudson et al., 1984
Ring-necked Pheasant	Acute Oral	Phenyl	65.0	LD$_{50}$ at 14 days	Mullins et al, 1977
	Acute Oral	Methyl	11.5	LD$_{50}$ at 14 days	Hudson et al., 1984
	Acute Oral	Ethyl	11.5	LD$_{50}$ at 14 days	Hudson et al., 1984
	Dietary at 15 day	Ethyl	112.0	LD$_{50}$	Hill, 1981; Spann, 1972
	Dietary at 28 day	Ethyl	37.4	LD$_{50}$	Hill, 1981; Spann, 1972
	Dietary at 70 day	Ethyl	4.2	LD0	Hill, 1981; Spann, 1972
	Dietary at 70 day	Ethyl	12.5	LD$_{50}$	Hill, 1981; Spann, 1972
Rock Dove	Acute Oral	Ethyl	22.8	LD$_{50}$ at 14 days	Hudson et al., 1984

GUIDELINES

Table 5.10: Proposed Protection Criteria for mercury in relation to protection of birds [After Eisler, 1987]

Species	Parameter	Value
Bird sp.	Daily Dose	<0.640 mg/kg body weight
Bird sp.	Diet	0.050-<0.100 mg/kg FW
Bird sp.	Egg	1.30-2.00 mg/kg FW
Mallard	Egg	<0.90 mg/kg FW
Ring-necked Pheasant	Egg	<0.90 mg/kg FW
Bird sp.	Feather	<5.00 mg/kg FW

D-5.5 2-Methyl fluorene (incineration source)

D-5.5.1 Vegetation

No information available

D-5.5.2 Earthworms

No information available

D-5.5.3 Cattle

No information available

D-5.5.4 Field Voles

No information available

D-5.5.5 Red Fox

No information available

D-5.5.6 Red-tailed Hawk

No information available

D-5.6 Nitrogen Oxides (Incineration source)

Nitrogen oxides can act on the terrestrial environment through either direct effects or through long range transport, deposition and indirect effects. The following table presents data from the scientific literature pertaining to direct effects of NOx on plants.

Table D-5.11:Direct Effects of Nitrogen Oxides on Plants

Species	Exposure	Concentration		Effect	Reference
		ppm	mg/m3		
Pinto bean	10 and 19 days	0.33	0.62	Appearance change without necrosis	Taylor et al, 1966
Broad bean	1 hour	1.0	1.9	Swelling of thylakoids chloroplasts	Wellburn, 1972
Tomato	37 days	0.25	0.47	Premature sencesence	Spierings, 1971
Tomato	10 and 22 days	0.11	0.21	Reduced foliage weight	Taylor et al, 1966
Tomato	128 days	0.25	0.47	Reduced growth, reduced fruit yield (22%), size (12% and number (11%)	Spierings, 1971
Tomato	10 days	0.5	0.94	Plants taller but smaller stem diameter	Spierings., 1971
Navel orange	35 days	0.50	0.9	Chlorosis and severe defoliation	Thompson, 1970
Navel orange	290 days NO_2	0.06	0.12	Increased fruit drop, reduced number and weight of fruit	Thompson, 1970
Citrus	4 to 8 hr	200	376	Rapid wilting and leaf abscission	MacLean et al, 1968
Citrus	1 hr	250	470	Rapid wilting and leaf abscission	MacLean et al, 1968
Potato	1 hr	100	188	Flower abscission	Czech et al., 1972
Oat	1 hr	100	188	Lodging	Czech et al., 1972
Oat	NO_2 for 90 min	0.6	1.1	Photosynthesis inhibition	Hill & Bennett, 1970
Oat	NO for 90 min	0.4	1.1	Photosynthesis inhibition	Hill &, Bennett, 1970
Alfalfa	NO for 90 min	0.4	0.8	Photosynthesis inhibition	Hill & Bennett, 1970
Alfalfa	NO_2 for 90 min	0.6	0.8	Photosynthesis inhibition	Hill & Bennett, 1970
Sugar Beet	1 hr NO_2	1000	1880	Root weight less than 1/3 of control	Czech et al, 1972
Sunflower	3 wks daylight	0.8	1.5	Increased plant height, dry weight of leaves, stems and roots	Faller

D-5.7 HCL (Incineration source)

D-5.7.1 Vegetation

Tomato plants exposed to 0.70 ppm for 10 hr/day for 2 days developed general interveinal chlorosis [Godish, 1970]. HCl concentrations in stems and roots of tomato and chrysanthemum plants fumigated with 5 ppm for 2 hours were not affected suggesting that any residues were retained in the leaves. [Shriner, 1969, Shriner et al., 1969]. In Pinto bean fumigated with HCl, the chloride ions accumulated in the leaves in proportion to the level of exposure [Endress, 1979a]. The ions tended to accumulate in the cell vacuoles of treated leaves.

The age of bean foliage influences sensitivity to HCl [Endress, 1979b]. Leaves younger than 9 days were resistant to 20 minute exposures of 25.6 mg HCl/ m^3.

Studies using bean and spinach subjected to gaseous fumigations of HCl were found to be injured at high rates of the gas (18.5 mg/m^{3}for 24 hours)^3[Endress, et al., 1982]. Photosynthesis and respiration were affected in proportion to leaf injury. Chloroplasts showed structural damage following exposure to the gas however the damage was at least partially reversible with time following cessation of the exposure [Endress et al., 1979]. Enzymatic activity (ribulose biphosphate carboxylase) in the leaves was stimulated in exposed leaves that did not sustain injury but was reduced in leaves that were injured. Damage was apparently related to direct acidification of the tissues.

Toxicity Values

Chronic effects of HCl gas can be expected on plants at 0.40 ppm while acute effects occur at 1-20 ppm [Anonymous, 1984]. Plants are generally more sensitive than animals.

Table D-5.12: Selected concentrations of HCl which can injure plants [Anonymous, 1984]

Species	Concentration	Effect
Pinto bean	6 mg/m^3 for 20 minutes	15% injury to leaf
Marigold	27.3 mg/m^3 for 20 min	Injury to leaves
Spinach	30.4 mg/m^3 for 20 min	Injury to leaves
Radish	13 to 27 mg/m^3 for 20 min	Visible injury at higher temperatures and light
Tomato	0.7 ppm 10 hr/d for 2 days	Interveinal chlorosis
	0.40 ppm 8 hr/d for 6 days	No visible injury

D-5.7.2 Earthworms

No information available

D-5.7.3 Cattle

The strong corrosive properties of hydrochloric acid can damage exposed animal tissues; however, fatalities from this are rare. The toxicity threshold for extended exposure for livestock is 1500 mg/L [Todd, 1970] in drinking water.

D-5.7.4 Field Voles

Various concentrations of HCl as vapours or aerosols have been reported as affecting rats [Anonymous, 1984]. These concentrations ranged as high as 60,100 mg/m^3 to reach an LD_{50} in 5 minutes. The lowest and more realistic exposure is 3124 mg/m^3 for 1 hour which was determined to be the lowest concentration causing death. Mice are more sensitive than rats as the HCl lowest concentration causing death was 1690 g/m^3 for 1 hour [Anonymous, 1984].

D-5.7.5 Red Fox

No information available

D-5.7.6 Red-tailed Hawk

The only information available for birds involved Rock Doves (pigeons) [Patty, 1981. Cited in Anonymous, 1984]. They experienced irritation of eyes and nose and slightly diminished haemoglobin concentrations under exposure conditions of 150 mg/m^3 HCl for 6 hours daily for 60 days. The birds also experienced "slight unrest".

D-5.8 Bromodichloromethane (Landfill source)

D-5.8.1 Vegetation

No information available

D-5.8.2 Earthworms

No information available

D-5.8.3 Cattle

No information available

D-5.8.4 Field Voles

No information available

D-5.8.5 Red Fox

No information available

D-5.8.6 Red-tailed Hawk

No information available

D-5.9 1,2-Dibromoethane (Landfill source)

D-5.9.1 Vegetation

Response of higher plants to 1,2-Dibromoethane [After Rannug, 1980}

Species	Response	Exposure	Reference
Barley	Chromosomal aberration	Not given	Ehrenberg, 1974
Tradescantia	Pink mutations in stamen hairs	5 ppm	Ma, 1978
Onion	No mutagenic effect	Not given	Kristoffersson, 1974

D-5.9.2 Earthworms

No information available

D-5.9.3 Effects of Exposure of Cattle to 1,2-Dibromoethane

Effects of 1,2-Dibromoethane in cattle [after Rannug, 1980]

Response	Exposure	Reference Cited
Abnormal shape and decreased motility of sperm, recovery to normal after 3 months	4. mg/kg 1,2-DBE from 4 days of age then every second day from 12 months old	Amir, 1973, 1975, 1977

D-5.9.4 Effects of Exposure of Field Voles to 1,2-Dibromoethane

Effects of 1,2-Dibromoethane on rodents

Species	Response	Exposure Concentration	
Rat	Single oral LD_{50} (female)	0.117 g/kg	Verschueren, 1983
	LD_{50} repeated inhalation	50 ppm, 7 hr/d, 5 day/week, 6 months	Verschueren, 1983
	Max exposure with no effects	50 ppm for 7 hrs	Verschueren, 1983
	No liver necrosis	75-100 mg/kg body weight, stomach intubation	Rannug, 1980
	93% survival at 52 weeks but with 42% with tumors	Inhalation 5 ppm 7 hr/day 5 days/week	Rannug, 1980
	Increased mortality at 15 mo. (30 vs 1 male; 9 vs 3 female)	Inhalation, 20 ppm for 18 months	Wong, 1982
	Increased tumors at 18 mos (25 vs 8 male; 29 vs 7 female)	Inhalation, 20 ppm for 18 months	Wong, 1982
	Mitotogenic	75-100 mg/kg body weight, stomach intubation	Rannug, 1980
	80% animals developed gastric carcinomas	40 mg/kg/day for 53 weeks via gastric untubation	Rannug, 1980
	Sub fertility (3rd week) and infertility (4th week)	100 mg/kg, i.p.	Rannug, 1980
	No fertility effects	1 g/kg orally for 5 days	Rannug, 1980
Guinea Pig	Single oral LD_{50}	0.110 g/kg	Verschueren, 1983
Rabbit	Single oral LD_{50} (female)	0.055 g/kg	Verschueren, 1983
Mouse	Single oral LD_{50} (female)	0.420 g/kg	Verschueren, 1983
	Formation of gastric carcinomas	60 mg/kg for 53 weeks via gastric intubation	Rannug, 1980
	No effect	18 and 900 mg/kg body weight, i.p.	Rannug, 1980
	No effect	50-100 mg/kg body weight, oral	Rannug, 1980

D-5.9.5 Effects of Exposure of Red Fox to 1,2-Dibromoethane

No information available

D-5.9.6 Effects of Exposure of Red-tailed Hawk to 1,2-Dibromoethane

Information for effects of 1,2-Dibromoethane in chickens		
Response	Exposure Concentration	Reference Cited
Irreversible cessation of egg production	50 g/day standard ration + 50 g supplement with 1,2-DBE at 180 ppm	Rannug, 1980
Number and weight of eggs reduced within 12 weeks	1,2-DBE in diet at 5-7.5 ppm	Rannug, 1980
Single oral LD_{50}	0.079 g/kg	Verhueren, 1983

D-5.10 1,1-Dichloroethane (Landfill source)

D-5.10.1 Vegetation

No information available

D-5.10.2 Earthworms

No information available

D-5.10.3 Cattle

No information available

D-5.10.4 Field Voles

1,1-Dichloroethane in rats and mice is mostly excreted unchanged through respiration.[Fawell, 1988]

Species	Response	Exposure Concentration	Reference
Rat	Inhalation LD_{50}	17,300 ppm for 2 hr	Verschueren, 1983
	Maximum tolerated dose	700 mg/kg	Fawell, 1988
	No effects	500 ppm 6 hr/day, 5 day/wk for 13 weeks	Fawell, 1988

Species	Response	Exposure Concentration	Reference
Rat (con't)	Inconclusive tumour study	Up to 764 and 950 mg/kg/day (male & female) respcctively by gavage for 78 weeks	Fawell, 1988
	No teratogenicity but bone ossification delayed	to 6000 ppm on days 6-15 of gestation	Fawell, 1988
Guinea pig	No effects	500 ppm 6 hr/day, 5 day/wk for 13 weeks	Fawell, 1988
	No effects	500 ppm 6 hr/day, 5 day/wk for 13 weeks followed by 13 weeks at 1000 ppm	Fawell, 1988
Cat	No effects	500 ppm 6 hr/day, 5 day/wk for 13 weeks	Fawell, 1988
	Nephrotoxicity and elevated blood urea	500 ppm 6 hr/day, 5 day/wk for 13 weeks followed by 13 weeks at 1000 ppm	Fawell, 1988
Mouse	Maximum tolerated dose	1800 mg/kg	Fawell, 1988
	No effects	500 ppm 6 hr/day, 5 day/wk for 13 weeks	Fawell, 1988
	Inhalation LD_{50}	16,000 ppm for 8 hr	Verschueren, 1983
	No effect	1000 ppm 5 x 6 hr/w, 3 months	Verschueren, 1983
	Inconclusive tumour study	Up to 2885 and 3331 mg/kg/day (male & female) respcctively by gavage for 78 weeks	Fawell, 1988
Rabbit	No effects	500 ppm 6 hr/day, 5 day/wk for 13 weeks	Fawell, 1988
	Tolerated	500 ppm 6 hr/day, 5 day/wk for 13 weeks followed by 13 weeks at 1000 ppm	Fawell, 1988

D-5.10.5 Effects of Exposure of Red Fox to 1,1-Dichloroethane

Dogs exposed to either 500 or 1000 ppm 1,1-Dichloroethane for 7 hr/day for 5 day/wk for 6 months showed no exposure related effects [Fawell, 1988].

D-5.10.6 Effects of Exposure of Red-tailed Hawk to 1,1-Dichloroethane

No information available

D-5.11 1,2-Dichloroethane (Landfill source)

D-5.11.1 Effects of Exposure of Vegetation to 1,2-Dichloroethane

Effects of 1,2-Dichloroethane on higher vegetation [after Rannug, 1980]

Species	Response	Exposure	Reference Cited
Barley	Chlorophyll mutation	Not given	Ehrenberg, 1974
Onion	Toxic	Low concentrations	Kristoffersson, 1974
	Induced c-mitosis	Low concentrations	Kristoffersson, 1974

D-5.11.2 Earthworms

Contact LC_{50} = 14 $\mu g/cm^2$ [Neuhauser et al, 1985]

D-5.11.3 Cattle

No information available

D-5.11.4 Field Voles

1,2-Dichloroethane is extensively metabolized in rat tissues. Radioactively-labelled residues accumulate in the liver and kidney regardless of the exposure route [Reitz, 1982]. About 85% of the dose is excreted in the urine within 48 hours with respiration and feces accounting for most of the remainder.

Genotoxicity of 1,2-Dichloroethane in mice is highly dependent upon the exposure route [Storer, 1984].

Species	Response	Exposure Concentration	Reference
Rat	Inhalation LC_{50}	12,000 ppm 31.8 min 3000 ppm, 165 min 1000 ppm 432 min	Verschueren, 1983
	Inhalation survival	100 ppm for 7 hr/day, 5 day/wk over many months	Verschueren, 1983

Species	Response	Exposure Concentration	Reference
Rat (con't)	No specific tumor type	Inhalation for 7 hr/day, 5 day/wk at 5,10,50 or 150 ppm after 104 weeks	Verschueren, 1983
	Oral LD_{50}	0.68 g/kg	Verschueren, 1983
	Oral LD_{50}	670 mg/kg	Verschueren, 1983
	Slight increase in lever fat	Diets with up to 500 ppm equal to 40 mg/kg body weight	Alumot, 1976
	Induced tumors	150 mg/kg/day by gavage	Reitz, 1982
Rabbit	Tolerated	200 ppm for several months	Fawell, 1988
Monkey	Tolerated	200 ppm for several months	Fawell, 1988
Guinea pig	inhalation survival	100 ppm for 7 hr/day, 5 day/wk, over many months	Verschueren, 1983
Mouse	Oral LD_{50}	489 mg/kg	Verschueren, 1983
	No specific tumor type	Inhalation for 7 hr/day, 5day/wk at 5,10,50 or 150 ppm after 104 weeks	Verschueren, 1983
	Induced tumors	150 mg/kg/day by gavage	Reitz, 1982
	Threshhold increased serum ATT	400 mg/kg	Storer, 1984
	Threshold increased serum IDH and AAT	500 mg/kg i.p.	Storer, 1984
	Threshold increased serum IDH and BUN	500 ppm for 4 hours	Storer, 1984
	80 % mortality	1000 ppm for 4 hours	Storer, 1984
	No effect on several reproduction and development indicators	Up to 0.29 mg/ml in drinking water in multi-generation study	Lane, 1982

D-5.11.5 Red Fox

Dogs displayed slight changes in liver structure when exposed to 400 ppm 1,2-Dichloroethane for 8 months [Fawell, 1988]. Clouding of dog corneas was reversible within 10 hours of subcutaneous administration of undiluted 1,2-Dichloroethane at 0.9 mg/kg body weight.

D-5.11.6 Red-tailed Hawk

The only available avian studies indicated that chickens fed diets containing 250 or 500 ppm 1,2-Dichloroethane for 2 years showed no effects on growth, fertility and blood components

[Alumot, 1976; WHO, 1987]. At the higher dose, fewer eggs were produced and egg weight decreased at both dosage levels at 4 months. A tolerance of 100 ppm in food and a daily intake of 5 mg/kg body weight were proposed.

D-5.12 1,1-Dichloroethylene (Landfill source)

D-5.12.1 Vegetation

No information available

D-5.12.2 Earthworms

No information available

D-5.12.3 Cattle

No information available

D-5.12.4 Field Voles

Verschueren {1983] reported slight nose irritation and normal organs in rats exposed to 200 ppm 1,1-Dichloroethylene for 20 6-hr exposures. In chronic oral biassays on rats significant injury to kidneys and liver occurred at 50 ppm or, when converted, 9 mg/kg/day 1,1-Dichloroethylene [Quast et al, 1983].

D-5.12.5 Red Fox

Beagle dogs treated for 97 days at rates of 6 to 25 mg/kg b.w./day in gelatin capsules showed no effect of 1,1-Dichloroethylene [Quast et al, 1983].

D-5.12.6 Red-tailed Hawk

No information available

D-5.13 Ethyl mercaptan (Landfill source)

D-5.13.1 Vegetation

No information available

D-5.13.2 Earthworms

No information available

D-5.13.3 Cattle

No information available

D-5.13.4 Field Voles

Natural product in urine of rabbits consuming cabbage [Budavari, 1989]

D-5.13.5 Red Fox

No information available

D-5.13.6 Red-tailed Hawk

No information available

D-5.14 Methyl mercaptan (Landfill source)

D-5.14.1 Vegetation

Methyl mercaptan has been reported as a natural component of radish roots and emitted by certain fungi (*Penicillium brevicaule*) [Budavari, 1989]

D-5.14.2 Earthworms

No information available

D-5.14.3 Cattle

No information available

D-5.14.4 Field Voles

LC_{50} for rats in air = 10,000 ppm [Budavari, 1989]

D-5.14.5 Red Fox

No information available

D-5.14.6 Red-tailed Hawk

No information available

D-5.15 Methylene chloride (Landfill source)

D-5.15.1 Vegetation

No information available

D-5.15.2 Earthworms

Contact LC_{50} = 19 $\mu g/cm^2$ [Neuhauser et at, 1985]

D-5.15.3 Cattle

No information available

D-5.15.4 Field Voles

Methylene chloride is metabolized by rats and the biochemical pathways are dose-dependant [McKenna, 1981]. Radioactively-labelled residues were mainly found in the liver and kidney. Un-metabolized methylene chloride as well as labelled CO and CO_2 metabolites was excreted during respiration. [McKenna, 1981 & 1982].

Response of rodents to methylene chloride

Species	Response	Exposure	Reference
Rat	LD_{50}	1600 mg/kg	Neuhauser, 1985
	LD_{50}	3000 mg/kg body weight oral	WHO, 1984
	88% metabolized in 48 hours	1 mg/kg body weight by gavage single dose	McKenna, 1981
	28% metabolized in 48 hours	50 mg/kg body weight by gavage single dose	McKenna, 1981
	Modified behaviour of young born to exposed mothers	4500 ppm before and/or during gestation	Bornschien, 1980

Response of rodents to methylene chloride			
Species	Response	Exposure	Reference
	No teratogenicity	4500 ppm before and/or during gestation	Hardin, 1980
	LC_{50} - 2 hours	Inhalation 79,000 mg/m^3	WHO, 1984
	LC_{50} - 6 hours	Inhalation 52,000 mg/m^3	WHO, 1984
	No effect on behaviour, blood and urine chemistry and estrus cycle	Drinking water at 125 mg/l for 13 weeks	WHO, 1984
	Increase in haematocrit, haemogobin and red blood cells; Decreased BUN, protein and cholesterol	Drinking water 136 to 263 mg/kg body weight per day for 24 months	WHO, 1984
	No significant increase in tumor formation	Drinking water, 263 mg/kg body weight per day for 24 months	WHO, 1984
Mouse	LC_{50} - 2 hours	Inhalation 51,500 mg/m^3	WHO, 1984
	LC_{50} - 6 hours	Inhalation 49,100 mg/m^3	WHO, 1984
	LC_{50} - 7 hours	Inhalation 51,500 mg/m^3	WHO, 1984
	Marginal increase in sarcomas in males	Drinking water 235 mg/kg	WHO, 1984
Rabbit	Carboxyhemoglobin	1270 to 11520 ppm	Roth, 1975
Guinea pig	LC_{50} - 6 hours	Inhalation 40,200 mg/m^3	WHO, 1984

D-5.15.5 Red Fox

Effects of methylene chloride on dogs		
Response	Exposure	Reference
LD_{50}	3000 mg/kg bodyweight as an oral dose	WHO, 1984
Central nervous system disorders	Inhalation 34,700 mg/m^3 for 7 hr/day, 5 days per week for 6 months.	WHO, 1984
Central nervous system disorder leading to narcosis	Inhalation 13,900 to 20,800 mg/m^3 after 2-6 hours	WHO, 1984

D-5.15.6 Red-tailed Hawk

No information available

D-5.16 1,1,2,2-Tetrachloroethane (Landfill source)

D-5.16.1 Vegetation

No information available

D-5.16.2 Earthworms

No information available

D-5.16.3 Cattle

No information available

D-5.16.4 Field Voles

Rats and mice were exposed experimentally to 1,1,2,2-Tetrachloroethane by gauvage at rates of 43 to 108 mg/k/day (rats) and 142-282 mg/kg/day (mice) [NCI, 1978]. Fifty percent of female rats died within five weeks at the high doses but mortality was not related to dose in male rats and no significant incidence of neoplasms was noted.. Mortality of mice was significantly related to dose as was the incidence of hepatocellular carcinomas

Paolini et al. [1992] determined the LD$_{50}$ to be 1476 mg/kg as a single dose (i.p.) based on studies using 300 or 600 mg/kg TTCE which is equivalent to 20 and 40% of the LD$_{50}$. The study determined that the TTCE affected hepatic cytochrome P-450 and and associated enzyme systems. They concluded that TTCE hepatotoxicity was related to *in vivo* lipid peroxidation.

Effects of 1,1,2,2-Tetrachloroethylene on rodents [after Fawell, 1988]

Species	Response	Exposure	Reference Cited
Mouse	LD$_{50}$	8.85 g/kg, Oral	Torkelson, 1982
	Increased mortality due to streptococcal infection	Inhalation, 50 ppm for 3 hours	Aranyi, 1986
	Fatty degeneration; No kidney effects or cell degeneration	Inhalation, at 200 ppm for 4 hrs/day, 6 days/week up to 8 weeks	Kylin, 1965

Effects of 1,1,2,2-Tetrachloroethylene on rodents [after Fawell, 1988]

Species	Response	Exposure	Reference Cited
	No chromosomal aberrations in bone marrow	Single, repeated or long-term exposures *in vivo*	Cerna, 1977
	Increase in hepatocellular carcinomas	500 or 100 mg/kg body weight by gavage, 5 days/week for 78 weeks	NCI, 1977
	Increased incidence of liver tumors	Inhalation at 100 or 200 ppm for 2 years	USNTP, 1986
Rat	LD$_{50}$	2.60 g/kg Oral	Torkelson, 1982
	LD$_{50}$	Oral in drinking water 3.84 g/kg male, 3.00female	Haynes, 1986
	Reduced body weight, no deaths; no haematology or urinary changes	Oral in drinking water 400 or 1400 mg/kgfemale, 1400 g/kg male	Haynes, 1986
	Remained conscious	Inhalation at 2000 ppm several hours	Rowe, 1952
	Unconsciousness	Inhalation at 3000 ppm in few hours	Rowe, 1952
	Unconsciousness	Inhalation at 6000 ppm, few minutes	Rowe, 1952
	Loss of coordination & balance plus CNS depression	Inhalation at 1600 ppm for 183 days	Rowe, 1952
	No effects	Inhalation at 400 ppm 7 hr/day for 183	Rowe, 1952
	No chromosomal aberrations in bone marrow	Single, repeated or long-term exposures *in vivo*	Cerna, 1977
	No increase in liver tumours	500 or 100 mg.kg body weight by gavage, 5 days/week for 78 weeks	Weisburger, 1977
	No increase in tumours	Inhalation at 315 of 602 ppm for 6 hr/day, 5 days/week for 12 months	Rampy, 1977
	Increased incidence of lukemia	Inhalation at 200 or 400 ppm for 2 years	USNTP, 1986
Guinea pig	No ill effects	Inhalation at 179 ppm 7 hr/day for 250 days	Rowe, 1952
	Increased liver weight & fat degeneration, Decreased hepatic ATP	Inhalation at 400 ppm 7 hr/day for 183 days	Rowe, 1952

Effects of 1,1,2,2-Tetrachloroethylene on rodents [after Fawell, 1988]

Species	Response	Exposure	Reference Cited
Rabbit	Threshold cardiotoxicity	6.5 mg/kg i.v. injection	Kobayashi, 1981
	No ill effects	Inhalation at 400 ppm 7 hr/day for 183 days	Rowe, 1952
	Loss of coordination & balance plus CNS depression	Inhalation at 255 ppm for 183 days	Rowe, 1952
Gerbil	Altered fatty acid composition in brain	Inhalation at 120 ppm for 12 months	Kryklund, 1984

D-5.16.5 Red Fox

Effects of 1,1,2,2-Tetrachloroethylene on dogs and cats [after Fawell, 1988]

Species	Response	Exposure	Reference Cited
Dog	Threshold cardiotoxicity	6.5 mg/kg i.v. injection	Kobayashi, 1981
Cat	Threshold cardiotoxicity	24.0 mg/kg i.v. injection	Kobayashi, 1981

D-5.16.6 Red-tailed Hawk

No information available

D-5.17 1,1,2-Trichloroethane (Landfill source)

D-5.17.1 Vegetation

No information available

D-5.17.2 Earthworms

Contact LC_{50} = 42 μg/cm^2 [Neuhauser et at, 1985]

D-5.17.3 Cattle

No information available

D-5.17.4 Field Voles

An LD_{50} dose for rats (single oral) was determined to be 10.3 to 12.3 g/kg 1,1,1-Trichloroethane [Halling-Sorensen, 1990]. No statistically significant or treatment-related effects could be detected in mice dosed (gauvage) with the chemical at rates of 2807 to 5615 mg/kg body weight, 5 days per week for 78 weeks although various neoplasms could be detected in all treatment groups [NCI, 1977]. Rats showed similar responses under similar exposures at rates of 750-1500 mg/kg [NCI, 1977].

D-5.17.5 Red Fox

No information available

D-5.17.6 Red-tailed Hawk

No information available

D-5.18 **Trichloroethylene** (Landfill source)

D-5.18.1 Vegetation

No information available

D-5.18.2 Earthworms

Contact LC_{50} = 105 μg/cm^2 [Neuhauser et al, 1985]

D-5.18.3 Cattle

No information available

D-5.18.4 Field Voles

Mice exposed (inhalation) to Trichloroethylene were found to metabolize over 98% of the chemical within 50 hours [Stott, 1982]. The primary route of excretion (75%) occured via the urine. The effects in rats was similar but a smaller percentage was excreted in the urine.

Effects of Trichloroethylene on rodents [mainly after WHO, 1985]

Species	Response	Exposure	Reference cited
Rat	LD_{50}	Oral 4,920 mg/kg	Anonymous. 1987

Effects of Trichloroethylene on rodents [mainly after WHO, 1985]			
	LD_{50}	4,920 mg/kg body weight, oral	Smyth, 1969
	LD_{50}	2,725 mg/kg body weight, i.p.	Rigaud, 1977
	LC_{low}	Inhalation 8,000 ppm/4 hours	Anonymous. 1987
	LC_{50}	Inhalation 12,500 ppm (66.9 mg/l) 4 hours	Siegel, 1971
	No effect	Inhalation, 3,000 mg/m^3 for 7 hours	Adams, 1951
	LC_{100}	Inhalation, 2,500 mg/ml for 4 hours	Adams, 1951
	Increased activity by 25%	Inhalation, 30 ppm continuous at 30 days	Kjellstrand, 1983
	Increased liver weight	Inhalation, 150 ppm continuous at 30 days	Kjellstrand, 1983
	No mortality, haematology, histology of major organs	Inhalation, 730 mg/m^3 for 8 hr/day, 5 days/wk for 6 weeks	Prendergast, 1987
Mouse	Completely metabolized	Inhalation to 3,240 mg/m^3 (600 ppm)	Anonymous, 1985
	LD_{50}	2,400 mg/kg body weight, oral in 24 hr	Smyth, 1969
	Minimum daily dose to induce peroxisomes	0.4 mmol/kg, oral	Larson, 1992
	Minimum daily dose to induce liver tumors	7.6 mmol/kg, oral	Larson, 1992
	LD_{50}	1,200 mg/kg body weight, i.p. in 24 hr	Aviado, 1976
	LD_{50}	3,000 mg/kg body weight, i.p. in 14 days	Gradiski, 1974
	LD_{50}	1,440 mg/kg body weight, subcutaneously	Smyth, 1969
Mouse	LD_{50}	34 mg/kg body weight, intravenous	NIOSH, 1977
	No mortality	Drinking water up to 5.0 mg/ml TCE equivalent to dose of 750 mg/kg females and 1250 males	Tucker, 1982
	Increased liver and kidney weight	Drinking water with range of TCE from 0.1 to 5.0 mg/ml	Tucker, 1982
	Slope acute toxicty = 0.86 in females and 0.72 in males	Drinking water with range of TCE from 0.1 to 5.0 mg/ml	Tucker, 1982

Effects of Trichloroethylene on rodents [mainly after WHO, 1985]			
	LC$_{50}$	Inhalation 8,450 ppm (45 mg/l) for 7 hr	Kylin, 1962
	LC$_{50}$	Inhalation 40 mg/l	Lazerev, 1977
	LC$_{low}$	Inhalation 3000 ppm (16 mg/l) 2 hours	Lazarev, 1929
Cat	LC$_{low}$	6,074 ppm (32.5 mg.l) 2 hours	Lehmann, 1936
Guinea pig	LC$_{100}$	37,000 ppm (198 mg/l) 0.67 hour	Von Oettingen, 1955
Rabbit	NOEL, haematology, histology of major organs	730 mg/m^3 8 hr/day, 5 days/wk for 6 weeks	Prendergast, 1967
	NOEL, haematology, histology of major organs	35 mg/m^3 for 90 days	Prendergast, 1967
	No maternal or embryo toxicity	2,700 mg/m^3 7 hr/day, 5 days/wk for 3 weeks pre and 2 weeks during gestation	Beliles, 1980
	NOEL, mortality	200 mg/m^3 8 hr/day, 5 days/week for 6 weeks	Adams, 1951
	LC	5000 ppm (27 mg/l) 14 hr	McCord, 1932

D-5.18.5 Red Fox

Effect of Trichloroethylene in dogs [after WHO, 1985]

Response	Exposure	Reference Cited
LD$_{50}$	5680 mg/kg body weight, oral	Christiansen, 1974
LD$_{50}$	2725 mg/kg body weight, i.p.	Klaassen, 1967
LD$_{low}$	150 mg/kg body weight, i.v.	Barsoum, 1934
NOEL,haematology, histology of major organs	730 mg/m^3 8 hr/ day 5 days/wk for 6 weeks	Prendergast, 1967
NOEL, haematology, histology of major organs	35 mg/m^3 for 90 days	Prendergast, 1967

D-5.18.6 Red-tailed Hawk

Effects of Trichloroethylene in chickens [after Bross, 1983 and WHO, 1985]

Response	Exposure	Reference Cited

67% dead embryos	1 to 25 μmol/egg injected	Bross, 1983
Abnormal beak, feet and feathers in 40-50% of chicks	1 to 25 μmol/egg injected	Bross, 1983
Increased embryo death	Eggs in 1% TCE	Fink, 1968
9 of 55 embryos with malformations	50-100 μmol/egg injected	Elovarra, 1979
LD$_{50}$ for eggs	50-100 μmol/egg injected into air space	Elovaara, 1979

D-5.19 Vinyl chloride (Landfill source)

D-5.19.1 Vegetation

No information available

D-5.19.2 Earthworms

No information available

D-5.19.3 Cattle

No information available

D-5.19.4 Field Voles

The effects of vinyl chloride have been summarized by Plugge [1977]. Vinyl chloride is excreted rapidly from rats. Following exposure of rats to 49 ppm VC for 65 minutes, 85% was excreted in 1 day (67% was excreted in the urine, 4% in feces and 14% respired as CO. At 75 hours, 1.6% occurred in liver, 3.6 % in skin, 0.2% in kidney and 7.6 % in carcass. Seventy five percent of a dose of 250 μg/kg inhaled VC was reported to be excreted in the urine. Eighty percent of intravenous VC was exhaled in 2 minutes, 99% exhaled in 1 hour [Warren, 1978]

The only information obtained relating carcinogenesis in rats to exposures of vinyl chloride was one report the noted that one rat developed liver angiosarcoma tumors after 50 weeks of feeding with 16.6 mg/kg VC [Warren, 1978]. All rats subjected to vapors of vinyl chloride at 3% volume given for 4 hr/day 5 days/ week for 12 months survived but oncogenic effects were noted a low as 250 ppm.

Rats given 100 or 300 mg/kg body weight per day via gavage for 6 days per week for 13 weeks showed reduced blood sugar (at 100 ppm), serum chemistry, urinary chemistry (glutamic oxaloacetc transaminase and glutamic-pyruvic transaminase), increased liver weight and reduced leucocyte counts (females only)

Table D-5.13: Toxicity of vinyl chloride inhaled by test animals [from Plugge, 1977]

Species	LD_{50}		LD_{100}	
	%v/v	mg/l	%v/v	mg/l
Rat	15.6	390	21	525
Guinea Pig	23.8	595	28	700
Rabbit	23.8	595	28	700
Mouse	11.75	293.7	4.75	118.8

D-5.19.5 Red Fox

Dogs exposed to 100,000 ppm (10% v/v) vinyl chloride for 4 hours developed cardia arrhythmia [Warren, 1978]

D-5.19.6 Red-tailed Hawk

No information available

D-5.20 Hydrogen Sulphide (Landfill source)

D-5.20.1 Vegetation

Vegetation can be injured by H_2S and it can accumulate in vegetation as sulphur [Thompson, 1976, 1978 & 1979; Oliva, 1976; McCallan, 1936; Cope, 1982]. Published data are presented below. It should be noted that the effects reported by McCallan [1936] cover concentrations in the parts per million range while those of Thompson are in the parts per billion range.

Response of vegetation species to H_2S fumigations			
Species	Effect	Exposure	Reference
Spinach	No injury, normal photosynthesis	0.6 mg/m^3 for 20 days	Oliva, 1976

Response of vegetation species to H_2S fumigations

Species	Effect	Exposure	Reference
	Decreased photosynthesis, increased transpiration,	1.2 mg/m^3 for 2 days	Oliva, 1976
Lettuce	Increased fresh weight	30 ppb continuous	Thompson, 1978
	Reduced yield	300 ppb continuous	Thompson, 1978
Alfalfa	No effects on yield	30 ppb continuous	Thompson, 1978
	Reduced yield	300 ppb continuous	Thompson, 1978
Grape	No effects on cane growth, increased leaf weight	30 ppb continuous	Thompson, 1978
	Reduced cane growth	300 ppb continuous	Thompson, 1978
Sugar beet	Increased leaf and root weight	30 ppb continuous	Thompson, 1978
	Reduced yield	300 ppb continuous	Thompson, 1978
Ponderosa Pine	No effects	30 ppb continuous	Thompson, 1978
	Foliar injury, increased leaf fall	300 ppb continuous	Thompson, 1978
Carnation	LOEC (foliar injury)	> 400 ppm for 4-5 hr	McCallan, 1936
Purslane	LOEC (foliar injury)	> 60 ppm for 4-5 hr	McCallan, 1936
Boston Fern	LOEC (foliar injury)	> 60 ppm for 4-5 hr	McCallan, 1936
Apple	LOEC (foliar injury)	> 60 ppm for 4-5 hr	McCallan, 1936
Cherry	LOEC (foliar injury)	> 60 ppm for 4-5 hr	McCallan, 1936
Strawberry	LOEC (foliar injury)	> 60 ppm for 4-5 hr	McCallan, 1936
Peach	LOEC (foliar injury)	60 ppm for 4-5 hr	McCallan, 1936
Coleus	LOEC (foliar injury)	60 ppm for 4-5 hr	McCallan, 1936
Pepper	LOEC (foliar injury)	50 ppm for 4-5 hr	McCallan, 1936
Rose	LOEC (foliar injury)	50 ppm for 4-5 hr	McCallan, 1936
Castor Bean	LOEC (foliar injury)	50 ppm for 4-5 hr	McCallan, 1936
Gladiolus	LOEC (foliar injury)	40 ppm for 4-5 hr	McCallan, 1936
Sunflower	LOEC (foliar injury)	40 ppm for 4-5 hr	McCallan, 1936
Nasturtium	LOEC (foliar injury)	40 ppm for 4-5 hr	McCallan, 1936
Buckwheat	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936
Cornflower	LOEC (foliar injury)	40 ppm for 4-5 hr	McCallan, 1936
Turkish tobacco	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936
Aster	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936

Response of vegetation species to H_2S fumigations

Species	Effect	Exposure	Reference
Soybean	LOEC (foliar injury)	40 ppm for 4-5 hr	McCallan, 1936
Kidney bean	LOEC (foliar injury)	<50 ppm for 4-5 hr	McCallan, 1936
Tobacco	LOEC (foliar injury)	<50 ppm for 4-5 hr	McCallan, 1936
Cucumber	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936
Salvia	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936
Poppy	LOEC (foliar injury)	50 ppm for 4-5 hr	McCallan, 1936
Tomato	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936
Clover	LOEC (foliar injury)	<40 ppm for 4-5 hr	McCallan, 1936
Radish	LOEC (foliar injury)	<20 ppm for 4-5 hr	McCallan, 1936
Calliopsis	LOEC (foliar injury)	<40 ppm for 4-5 hr	McCallan, 1936
Cosmos	LOEC (foliar injury)	<40 ppm for 4-5 hr	McCallan, 1936

Plant foliage can eliminate excess sulfur in the form of H_2S. The rates of H_2S production are low and dependent on plant species, light, concentration and sulphur form . The maximum emission rates reported by Wilson [1978] are indicated below. The importance of light in mitigating H_2S toxicity in rice was also investigated by Vamos [1972]. That study indicated that the free OH radicals produced during highter rates of photosynthesis protected the catalase and peroxidase enzymes and the glycolic acid oxidation reactions in plants and thereby overcame the toxicity of H_2S. In Monterey pine (*Pinus radiata*), about half of the radio-labelled sulphur take up in H_2S had been lost after 150 hours [Cope, 1982].

Maximum rates of H_2S emmisions from plant leaves [after Wilson, 1978]

Species	Rate*	Species	Rate
Pumpkin	6.78	Cotton	4.84
Squash	9.00	Corn	1.69
Cucumber	0.93	Soybean	1.69
Cantelope	9.16		

* Rates are reported as nmoles/min/leaf of 1-3 g fresh weight

D-5.20.2 Earthworms

No information available

D-5.20.3 Cattle

Hayden [1990] reported that cattle exposed to 20 ppm hydrogen sulphide for 7 h/d developed lethargy and discomfort.

D-5.20.4 Field Voles

Effects of hydrogen sulphide on rodents

Species	Response	Exposure	Reference
Mouse	LC_{50}	Inhalation, 24 mg/m^3 > 960 min	Verschueren, 1983
Rat	LC_{50}	Inhalation, 380 mg/m^3 >960 min	Verschueren, 1983
	Decreased cytochrome oxidase (15%)	4-hour exposure at 50 ppm	Kahn, 1990
	Decrease cytochrome oxidase (42%) and succinate oxidase (39%)	4-hour exposure at 200 ppm	Kahn, 1990
	Decreased viability of pulmonary alveolar macrophages (18%)	4-hour exposure at 560 mg/m^3	Kahn, 1991
	Increased blood taurine (30%)	50 ppm for 7 h/d for 15 days during pregnancy	Hannah, 1990
	Abnormal development of nervous system	50 ppm for 7 h/d for 15 days during pregnancy	Hannah, 1991
	Altered amino acid levels in developing nervous system	20 ppm for 7 h/d for 20 days during pregnancy	Skrajny, 1992
	Elevated blood glucose in mothers (36%)	20 ppm for 7 h/d for 20 days during pregnancy	Hayden, 1990
	Increased cholesterol in young	20 ppm for 7 h/d for 20 days during pregnancy	Hayden, 1990b
	Mortality	2317 mg/m^3 fir 3 minutes	Lopez, 1989
	160 % increase in number of cells in nasal fluid	4-hr exposure at 10 ppm	Lopez, 1987

	Exfoliation of nasal epithelial cells, lethargy	569 mg/m^3 for 4 hours	Lopez, 1988

D-5.20.5 Red Fox

Effect of hydrogen sulphide on dogs			
Species	Response	Exposure	Reference
Dog	Local irritation of eyes and throat, more severe reactions at increased concentrations	150 mg/m^3, many hours	WHO H_2S, 1981

D-5.20.6 Red-tailed Hawk

Effect of hydrogen sulphide on birds			
Species	Response	Exposure	Reference
Canary	Local irritation of eyes and throat, more severe reactions at increased concentrations	150 mg/m^3, many hours	WHO H_2S

For refernces, see section D-6.10.

D-6. RISK CHARACTERIZATION AND CONCLUSIONS

6.D-6. Risk Characterization of Chemical Emissions

The chemicals that are of potential for concern emitted by landfills and incinerators consist of both carcinogens and non-carcinogens. These two groups require different methods of characterizing the potential for risk to the environment from the presence of a specific chemical. For non-carcinogens (threshold chemicals) a concentration can often be determined below which effects have not been observed and would not be expected to occur. This assessment will, for non-threshold chemicals, attempt to compare the anticipated threshold concentrations or doses with those calculated by the models used. On the other hand, carcinogens might not have a threshold level below which effects don't occur; rather, the risk of an animal getting cancer increases with exposure to the chemical. Therefore, it is necessary to define what level of additional risk is acceptable to the species in question. For people, where the concern is with individuals, from 1 in 10,000 to 1 in 1,000,000 have been postulated as reasonable levels of risk. For animal populations, the philosophy is different. The death rate due to natural factors, such as predation, is often very high, and can overwhelm the potential effects of carcinogens present at low concentrations. Since most of the carcinogenistic effects are only seen later in the life of the animal, the individual has normally reproduced before the cancers develop, and hence the effect of low levels of carcinogens on populations is believed to often be insignificant. Animal populations can readily sustain cancer risks much higher than those deemed acceptable for individual humans. The state of New York [Newell et al., 1987] has used a risk of 1:100 as being the acceptable risk for non-human receptors. This would seem to be reasonable considering the factors mentioned above, as this risk level would not be expected to have any meaningful effect on populations.

For the purposes of this assessment, the risk characterization consists of both a comparison of projected concentrations or doses with those reported in the literature to have produced effects, and a calculation of risk due to the presence of carcinogens. It is assumed that if the additional risk from the presence of chemicals exceeds 1 in 100, there is need for concern for the particular population. Although the unit risks used in this assessment were developed for humans, much of the data on which they are based were derived from experiments on animals. The same unit risks that were used in the human health risk assessment were therefore used, except in situations where evidence exists that indicates they do not have relevance to other animals. There is an added degree of conservatism in this assessment in that the lifetimes of all the receptors in this study are considerably shorter than that of humans, and hence the opportunity would not be as great for the development of tumours.

D-6.1.1 Dioxins/Furans

Toxicity to birds

The data in the hazard assessment section indicates that the most sensitive endpoint for dioxins is likely to be toxicity to embryos in developing eggs of birds. The exposure data indicates that birds are likely to have the highest body burdens of dioxins and furans of any of the chosen receptors.

The CAPCOA model projects that the soil concentration for dioxin originating from the incinerator would be 0.000092 ng/g. When used in the worm-bird model, 0.0016 ng/g dioxins/furans would accumulate in earthworms, 0.25 ng/g would accumulate in the woodcock, and 0.00092 ng/g in eggs. When compared with the concentrations below which effects have not been observed (see table below), it is clear that the maximum possible concentrations are substantially below the potential effects level. If a factor is introduced that more realistically (i.e. < 1.0) reflects the transfer of dioxin from one trophic level to the next, the differences between the projected concentrations and effects levels would be even greater.

Table D-6.1: Comparison of modelled and biological effects levels for dioxins

Medium	Projected Conc.	Effect Level	
		Soil	Tissue
Soil	0.000092 ng/g	<0.010 ng/g*	-
Earthworm	0.0016 ng/g	5000 ng/g	>2200 ng/g
Woodcock	0.25 ng/g	-	15 ng/g
Egg	0.00092 ng/g	-	0.065 ng/g

*Criteria used by MOEE for soil, based on potential effects on songbird eggs. [Birmingham and Clark, 1993; MOEE, 1994]

Of the receptors covered by the modelling, the Red-tailed Hawk receives the largest body burden of dioxins, with a projected tissue concentration of 0.02 ng/g (20 ppt) from an ingestion dose of 6.14E-6 μg/kg/d and a dietary concentration of 6.0E-8 μ/g. These are all more than two orders of magnitude lower than the lowest effects concentrations, hence, no meaningful effects of dioxins emissions from the incinerator would be expected on hawks. Since body burdens for other receptors are lower and birds are among the most sensitive receptors,, there is little likelihood for adverse effects on terrestrial receptors.

D-6.1.2 Mercury

Toxicity to birds

The CAPCOA model projects that the soil concentration for mercury originating from the incinerator would be 148 μg/kg. When used in the worm-bird model, 217 ng/g mercury would accumulate in earthworms, and 4500 ng/g would accumulate in the woodcock. When compared with the levels that have been reported to have biological effects (see table below), the daily dose and dietary concentrations are seen to be lower than the potential effects levels. However, calculated whole body burdens for the birds are at concentrations where effects have been noted. Conservatism is built in to these estimates due to both the use of a 24 hr maximum ground level concentration and the fact that the studies from the literature are based on methyl mercury, which is considerably more toxic than inorganic mercury, and is taken up by the gut in much higher proportions than are other forms. In addition, the modelling for the hawk assumes that there is no loss of mercury from the tissue. Due to these assumptions it may appear reasonable to conclude that effects of mercury on bird populations would not be expected; however, a more detailed assessment would have to be conducted to be able to make a definitive statement regarding potential effects.

Table D-6.2: Comparison of modelled and biological effects levels for mercury

Medium	Parameter	Projected Conc.	Effect Level
Earthworm	Whole Body	217 ng/g	150000 ng/g
Hawk	Whole body	6700 ng/g	4300 ng/g
	Daily Dose	2 ng/g BW	640 ng/g BW
	Diet	18.5 ng/g	500 ng/g FW
Woodcock	Whole Body	4500 ng/g	4300 ng/g
	Daily Dose	185 ng/g BW	640 ng/g BW
	Diet	217 ng/g	500 ng/g FW

Toxicity to Mammals

From the modelling, the mammal that receives the largest dose of mercury from incinerator emissions is the field vole, which has a total ingestion dose of 8.2 μg/kg BW/d from a diet consisting of 0.024 μg Hg/g. The literaure indicates that effects have been found in rats at doses as low as 400 μg/kg BW/d and dietary concentrations of 2 μg Hg/g. Effects have been found in cats at ingestion rates of 46 μg/kg BW/d. The projected intakes, although approaching levels of concern, are still below doses that would be expected to produce meaningful effects on mammalian populations, especially considering the conservatism, as discussed above, that is built into this assessment.

D-6.1.3. 2-Methylfluorene

Data could not be found on the toxicity of 2-Methylfluorene to any of the receptors, nor is there data available on its normal background concentrations. It is not, therefore, possible to characterize the risk posed by this substance.

D-6.1.4 Benzene

Anticipated worst case ground level concentrations for benzene are 0.0056 $\mu g/m^3$ for an incinerator and 0.26 $\mu g/m^3$ for a landfill. The threshold of effects for vegetation occurs at about 3,000,000 μ/m^3. Acute toxic effects on animals do not occur until concentrations are in the grams/m^3 range; therefore, acute effects are not a concern for either landfills or incinerators. However, benzene is a carcinogen. The unit risk in humans for inhalation of benzene is 8.3E-06 (risk of leukemia per $\mu g/m^3$ in air). Since the modelled maximum 24 hr ground level concentration is 0.26 $\mu g/m^3$, the cancer risk from inhalation is 2.17E-06, on the basis of the assumptions made earlier, any potential effect on populations of the identified animal receptors would be imperceptible. This conclusion appears to be supported by animal studies (see Hazard Assessment) which do not show significant increases in cancers until air concentrations are in the hundreds of ppm range.

The unit risk for ingestion of benzene by humans is estimated in IRIS [1994] to be 0.029 per mg/kg/d; however, IRIS states that studies based on animal gavage would give a value approximately 5 times higher. It is therefore appropriate to use 0.145 per mg/kg/d as an appropriate oral slope factor for animals. Due to its high proportion of total dose coming from ingestion and to its low body weight, the red-tailed hawk is the most likely of the chosen receptors to exceed acceptable intakes. Its total dose is 5.2 μg/day, 4.7 of which is ingested. With a body weight of 1.2 kg, this gives 3.9 μg/kg/day as the ingestion dose. The risk of a hawk getting cancer from ingestion of benzene from the landfill would therefore be 5.7 E-04, assuming that it was always eating food obtained from the maximum point of impingement of the landfill emissions. As per the earlier discussion, this risk is unlikely to have any meaningful effect on populations in the area. Since maximum ground level concentrations from incinerators are lower than those from landfills, there is also unlikely to be any meaningful effect of benzene emissions on any of the receptors chosen.

D-6.1.5 Nitrogen Oxide Emissions

NOx is phytotoxic at high concentrations. Dispersion models indicate 24 hr POI values of 32 $\mu g/m^3$ for the incinerator. The ambient air quality criteria for NO_2 are 400 $\mu g/g$ for 1 hour and 200 $\mu g/g$ for 24 hours. Because the criteria were established at values which also protect vegetation, there should not be a measurable direct negative impact on vegetation provided that the projected POI concentration is met. This is supported by the data presented in the hazard assessment section.

Calculation of NO_X Emissions from Incinerator

NO_X Emission Rate = 19.04 g/sec
=1.645 x 10^6 g/day
=1,645 kg/day
@ 400 tonnes of MSW/day

Incineration produces 4.11 kg nitrogen oxides per tonne of garbage burned. For 6.6 million tonnes of garbage incinerated, a total of 27,000 tonnes of nitrogen oxides would be produced. Flaring of methane gas from the landfill would produce 500 tonnes of nitrogen oxides over 40 years. Although N_2O has a Global Warming Potential (GWP) of 270 that of CO_2, it is not thought to contribute significantly to the nitrogen oxide concentration of incinerator gases. NO_x, although a major contributor to the production of tropospheric ozone and to acidic depostion, is generally considered to be too short lived to be a contributor to global warming.

Eventually, most of the NO_x emissions are deposited back to earth, usually in precipitation. The nitrogen is converted to nitrates which are used a nutrients by plants. The nitrogen will act as a fertilizer and enhance the growth of plants in nitrogen deficient soils. In situations where soils are not nitrogen deficient, the amounts of nitrogen may be excessive and cause disruption of normal nutrient cycling in soil, including excellerating the rates of base cation losses due to leaching. The overall effect is to reduce forest productivity. Studies funded by the MOEE to determine target loadings for nitrogen suggest that nitrogen loadings may already be excessive for sensitive upland soils in southern Ontario, and, by extrapolation, for many northern Ontario soils. Any additional contributions of nitrogenous compounds to atmospheric loadings would appear to be undesirable, although additional field work and experimental studies are needed to confirm this and to assess the overall impact of NO_x.

In conclusion, from the perspective of maintaining the health of upland forests, current information indicates that the 50-fold difference in NO_x emissions between landfills and incinerators would render a largescale conversion to the use of incinerators as the less desirable option. However, this consideration would have to viewed in light of the potential beneficial effect that increased NO_x deposition could have for agricultural activities (reduced fertilizer cost).

D-6.1.6 HCl

The modelled maximum 24 hr ground level concentration is 2.3 $\mu g/m^3$ from the incinerator. Information from the hazard assessment section indicates that plants can be affected by HCl at concentrations as low as 0.4 ppm. (626 $\mu g/m^3$). Available information indicates that animals are considerably less sensitive to HCl than are plants; hence, it is unlikely that any direct effects of HCl on the terrestrial ecosystem would occur.

D-6.1.7 Methyl and Ethyl Mercaptan

Very little information was available on the toxicity of the mercaptans to terrestrial receptors. An LD_{50} for Methyl mercaptan in air for rats of 10,000 ppm (20,000 mg/m^3) indicates that the likelihood of any meaningful toxic effects is extremely remote at the projected 24 hr POI values of 2.52 $\mu g/m^3$ for Methyl mercaptan and 1.64 $\mu g/m^3$ for Ethyl mercaptan. The main consequence of the presence of these gases is an odour problem, which could not be assessed here.

D-6.1.8 Hydrogen Sulphide

The maximum ground level concentration of hydrogen sulphide in the vicinity of the landfill is 0.10 μg/m^3. Plants are the most sensitive receptor and exhibit effects at concentrations as low as 30 ppb (44 μg/m^3), about 2 orders of magnitude lower than those for which effects are seen in animals. Since this is more than two orders of magnitude higher than projected concentrations, there is unlikely to be any effect on the terrestrial system.

D-6.1.9 Bromodichloromethane

No information was obtained on the toxicity of Bromodichloromethane to any of the receptors. The receptor most likely to be at risk from exposure to Bromodichloromethane is the Red-tailed Hawk, which would, under worst case assumptions, ingest about 36 μg/kg/d and inhale 2.48 μg/kg/d

D-6.1.11 1,2-Dibromoethane

Of the receptors for which modelling was conducted, the Red-tailed hawk has the highest dose per unit body weight at 2.23 μg/kg/d ingested and .0097 μg/kg/d inhaled. The toxicity information indicates that effects are not observed until the mg/kg/d range is reached; hence, there is little chance of direct effects of this contaminant on wildlife.

D-6.1.12 1.1-Dichloroethane

Of the receptors considered, the Red-tailed hawk receives the highest dose of 1.1-Dichloroethane. Ingestion exposure to the Red-tailed hawk totals 110 μg/kg/d, which is the highest of any of the receptors and inhalation exposure is 7.0 μg/kg/d. The literature indicates that effects are not likely until ingestion exposures are in the hundreds of mg/kg/d or until concentrations in air are in the hundreds of ppm. The chances of adverse effects on wildlife are therefore extremely remote.

D-6.1.13 1,2- Dichloroethane

Of the receptors considered, the Red-tailed Hawk receives the highest dose of 1,2-Dichloroethane. Ingestion exposure to the Red-tailed Hawk totals 1.22 μg/kg/d, and inhalation exposure is 0.081 μg/kg/d. The literature indicates that effects are not likely until ingestion exposures are in the tens or hundreds of mg/kg/d or until concentrations in air are in the hundreds of ppm. A maximum daily intake rate of 5 mg/kg for birds was recommended in one study. There is therefore little possibility of adverse effects on wildlife from landfill emissions.

D-6.1.14 1,1-Dichloroethylene

Of the receptors considered, the Red-tailed Hawk receives the highest dose of 1,1-Dichloroethylene. Ingestion exposure to the Red-tailed Hawk totals 16.5 μg/kg/d, and inhalation exposure is 0.54 μg/kg/d. Although very little information is available in the literature, one study showed no effects on beagles ingesting 25 mg/kg/d, and another showed slight irritation on rats inhaling 9 mg/kg/d. These doses are three orders of magnitude higher than those modelled for the maximum GLC for landfills. Therefore there is little possibility of adverse effects on wildlife occurring from landfill emissions.

D-6.1.15 Methylene Chloride

Of the receptors considered, the Red-tailed Hawk receives the highest dose of Methylene chloride. Ingestion exposure to the Red-tailed Hawk totals 336 μg/kg/d, inhalation exposure is 16.5 μg/kg/d, and ground level concentrations are 11 $\mu g/m^3$. Toxicity information available indicates that effects from ingestion do not occur until exposure exceeds hundreds of mg/kg/d, and effects from inhalation do not occur until concentrations are in the thousands of mg/m^3 range. Again, the concentrations and doses resulting from the landfill appear to be about three orders of magnitude lower than those required to produce effects on wildlife.

D-6.1.16 1,1,2,2-Tetrachloroethane

Of the receptors considered, the Red-tailed Hawk receives the highest dose of 1,1,2,2-Tetrachloroethane. Ingestion exposure to the Red-tailed Hawk totals 2.8 μg/kg/d, inhalation exposure is 0.25 μg/kg/d, and ground level concentrations are 1.8 $\mu g/m^3$. The lowest concentration in air at which effects were seen in the literature was at 50 ppm (360 mg/m^3). This is more than five orders of magnitude higher than modelled air concentrations. For ingestion, 100 mg/kg/d produced an increase in hepatocellular carcinomas in mice, but not in rats. Again, this dose is more than four orders of magnitude higher than that encountered at the maximum GLC near a landfill. No effects on wildlife would be anticipated.

D-6.1.17 1,1,2-Trichloroethane

Of the receptors considered, the Red-tailed Hawk receives the highest dose of 1,1,2-Trichloroethane. Ingestion exposure to the Red-tailed Hawk totals 5.2 μg/kg/d, inhalation exposure is 0.097 μg/kg/d, and ground level concentrations are 0.065 $\mu g/m^3$. Very little toxicity information was available on 1,1,2-trichloroethane; however, one study showed no effects on mice at doses 5.6 g/kg/d, and another determined an LD_{50} for rats for a single oral dose of about 10 g/kg of 1,1,1-trichloroethane. These studies indicate that effects are highly unlikely at the concentrations encountered from landfills.

D-6.1.18 Trichloroethylene

Of the receptors considered, the Red-tailed Hawk receives the highest dose of Trichloroethylene. Ingestion exposure to the Red-tailed Hawk totals 17.6 μg/kg/d, inhalation exposure is 1.9 μg/kg/d, and ground level concentrations are 1.3 μg/m^3. The lowest ingestion dose found in the literature to produce an effect was 0.4 mmol/kg/d (53 mg/kg/d), more than three orders of magnitude higher than the anticipated dose for Red-tailed Hawk. Air concentrations required to produce effects are in the thousands of mg/m^3, more than five orders of magnitude higher than GLCs that were estimated. However, some studies where trichloroethylene was injected directly onto the developing embryo of chicken eggs found a high degree of sensitivity to the contaminant. In these situations, death of the embryo resulted from concentrations as low as 1 μmole/egg (approximately 0.0023 μg/kg in the egg). Results from such studies are extremely difficult to compare with anticipated doses to the hawk due to the difference in mode of exposure; however, they do indicate that additional special concern may be warranted for the developing embryos of birds. Since the anticipated concentration in the hawk is 10 μg/g, the possibility for concern does exist. A more detailed assessment of Trichloroethylene and other compounds to which this sensitivity may apply would be required in order to better determine the environmental significance of the presence of these compounds near landfills.

D-6.1.19 Vinyl Chloride

Of the receptors considered, the Red-tailed Hawk receives the highest dose of Vinyl chloride. Ingestion exposure to the Red-tailed Hawk totals 10.9 μg/kg/d, inhalation exposure is 1.16 μg/kg/d, and ground level concentrations are 0.077 μg/m^3. Although little directly useable data are available on the toxicity of vinyl chloride to the chosen receptors, the literature indicates effects being noted (tumor development) at an ingested dose as low as 16 mg/kg/d. This is three orders of magnitude higher than the anticipated doses, and meaningful effects would not be anticipated. LD_{50}s for inhalation are in the percentage range, and therefore, similarly, direct effects would not be anticipated.

D-6.1.20 Total Carcinogenicity

Table D-6.3 shows the calculation, using exposure-modelling based on annual average ground level concentrations, of the total risk,assuming additivity, of cancer for the compounds that may be of significance at the landfill. The ingestion dose is based on the Red-tailed Hawk, which receives the highest doses of all the receptors modelled.

Table D-6.3: Carcinogenic risk to Red-tailed Hawk from landfill emissions

Compound	Ingestion Dose (μg/kg/d)	Slope Factor (μg/kg/d)E-1	Risk
Benzene	0.467	2.90E-5	1.35E-7
Bromodichloromethane	3.91	6.20E-5	2.42E-4
1,2-dibromoethane	0.241	NA	
1,1-dichloroethane	19.8	9.10E-5	1.80E-3
1,2-dichloroethane	0.178	9.10E-5	1.62E-5
1,1-dichloroethylene	1.87	6.00E-4	1.12E-3
Methylene chloride	25.5	7.50E-6	1.91E-4
1,1,2,2-tetrachloroethane	0.333	2.00E-4	6.60E-5
1,1,2-trichloroethane	0.609	5.70E-5	3.47E-5
trichloroethylene	1.91	6.00E-6	1.15E-5
vinyl chloride	1.46	5.26E-4	7.68E-4
Total			4.24E-3

Not applicable - carcinogenic via inhalation.

The total risk from ingestion of landfill chemicals by a Red-tailed Hawk would be anticipated to be approximately 4.2 in 1000. Although this is under the 1 in 100 criteria referred to earlier, it is approaching a level of concern. It should be noted that a high degree of conservatism is built into this estimate due to the foraging area assumptions and the differences between the life-span of hawks and humans

D-6.1.21 Summary of Risk Characterization of Chemical Emissions

Table D-6.4 gives a numerical summary of the comparisons of effects concentrations with projected doses. Numerous assumptions have been used in this summary table. Most notable is the use of the lowest effects based number, or in some cases no effects number, that could be found that had any relevance to the receptor. In some cases the comparison used may be, for example, that of an effect on a mammal with the dose received by a hawk. The intent is to give the worst case situation based on actual data, given the lack of receptor specific data. Species to species extrapolation adjustments have not been made, as the means of comparison is already highly conservative.

Table D-6.4: Summary of Risk Characterization for Modelled Chemicals for Pathway of Greatest Concern

Incinerator Chemicals				
Chemical	Species Most Likely to be Affected	Effects Dose or Concentration	Projected Ingestion Dose or Concentration	Ratio of Projected: Effects
Benzene	Hawk	0.000145 per ng/g BW/d unit risk	1.94 ng/g BW/d	Cancer risk = 2.8E-4
Dioxins/Furans	Woodcock	15 ng/g in tissue	0.25 ng/g in tissue	0.017
Mercury	Hawk	4300 ng/g in tissue	6700ng/g in tissue	1.56
2-methyl fluorene	Hawk		1.29 ng/g BW/d	

Landfill Chemicals				
Chemical	Species Most Likely to be Affected	Effects Dose or Concentration	Projected Ingestion Dose or Concentration	Ratio of Projected: Effects
Benzene	Hawk	unit risk of 0.000145 per ng/g bw/d	3.9 ng/g bw/d	Cancer risk = 5.7E-4
Bromodichloromethane	Hawk		36 ng/g bw/d	
1,2-dibromoethane	Hawk	5,000 ng/g in diet	2.23 ng/g bw/d or 22 ng/g in diet	0.00045
1,1-dichloroethane	Hawk	760,000 ng/g bw/d (rat)	110 ng/g bw/d	0.007
1,2-dichloroethane	Hawk	5,000 ng/g bw/d	1.22 ng/g bw/d	0.004
1,1-dichloroethylene	Hawk	9,000 ng/g bw/d	16.5 ng/g bw/d	0.0018
Ethyl mercaptan	NA - odour problem			
Methyl mercaptan	NA - odour problem			
Methylene chloride	Hawk	136,000 ng/g bw/d	336 ng/g bw/d	0.0025
1,1,2,2-tetrachloroethane	Hawk	100,000 ng/g bw/d	2.8 ng/g bw/d	0.000028
1,1,2-trichloroethylene	Hawk	5,600,000 ng/g bw/d	5.2 ng/g bw/d	0.00000093
Trichloroethylene	Hawk	53,000 ng/g bw/d	17.6 ng/g bw/d	0.00033
vinyl chloride	Hawk	16,000 ng/g bw/d	10.9 ng/g bw/d	0.00068
HCl	Plants	626 $\mu g/m^3$	2.3 $\mu g/m^3$	.0037
Hydrogen Sulphide	Plants	44 $\mu g/m^3$	0.1 $\mu g/m^3$	.0023

D-6.2 Erosion

Erosion of materials from landfill sites is a potential problem that is now normally addressed in the design phase. If certain design restrictions are adhered to and a vigorous vegetative cover can be established, then soil erosion need not be a problem. The question of environmental impacts on terrestrial receptors from soil erosion therefore becomes one of the design of the facility and the maintainance of vegetative cover. For the purposes of this assessment, an assumption was made that the facility design is adequate to prevent any significant erosion (i.e. slopes are less than 4:1 (H:V), final cover of soil is of adequate quality to permit vigorous vegetative growth is sufficiently deep to allow root establishment and continued growth, gas collection systems are sufficient to prevent build-up of landfill gases in the soil to levels toxic to plants, and leachates are collected and are not permitted to seep out onto any slopes), and that there is sufficient control and care taken in the future to assure the health of the vegetative cover. The IWA documentation on the Durham site indicates that the required design features have been taken into account

Similar assumptions must also be made for the incinerator site. Although erosion from the site of the facility would not be expected to constitute any more of a problem than at a normal industrial site, provisions must be made in the design to assure that both the stockpiled ash and the landfill for the ash are adequately protected from erosional forces.

Since provision of adequate design and maintenance can result in erosion being of negligible concern for both landfilling and incineration, the terrestrial effects resulting from erosion should be no different for the two facilities, provided that the assumptions are met.

D-6.3 Drainage and Water Table Effects

A review of investigations conducted by the Phytotoxicology Section relating to landfills and incinerators (see Appendix 1) indicates that the most frequent cause of damage to terrestrial plants resulting from landfills is the alteration of drainage patterns and subsequent flooding of plant rooting areas. The plant roots can become starved of oxygen and the plant eventually dies. The situations where this has occurred are normally amenable to amelioration through simple engineering works. Proper consideration of drainage and potential effects on adjacent plants during the planning phases can prevent these effects of excess water from occurring. As such, for the purposes of this study, it must be assumed that the design of the drainage system is such as to prevent ponding of water on properties adjacent to the landfill. In this case, then, there would be no effect on plants or other existing habitats due to flooding. A similar assumption is also made for the incinerator facility.

Since landfills are often pumped dry of water for the period of operation and since they have leachate collection systems that collect large amounts of water and conduct it through treatment plants and off-site, they often result in a drawdown of the water table in the immediate vicinity. Most plants native to Ontario obtain their water from the soil and do not rely on the water table as a source of water; however, situations could arise, such as that of a pine forest on a sandy soil, where the supplementary water from the water table may be important to the forest in a dry year. A drawdown as significant as that modelled in the IWA report for Durham could produce additional stress for such vegetation. However, since, in most situations, including that of the Durham site, such effects are not anticipated, and since these effects would have to be accounted for in the case of an incinerator as well (both for the site and for the mono-fill for the ash) there should be no difference between the two types of facilities regarding the effect of water table drawdown on terrestrial species.

Air Drainage

Large perimeter berms around landfills act as a barrier to the flow of air. Under slope conditions that permit the ponding of cold air due to the blockage of air movement, plants in the susceptible area can be subject to increased frequency of early fall and late spring frosts. For plants that are sensitive to frost, such as tender fruit crops, these additional frosts can significantly affect production. Landfills can be designed to prevent any such effects on adjacent properties. The EE11 site is located sufficiently close to valleys and drainage channels such that air drainage should adequate to prevent these types of adverse effects.

D-6.4 Effects of Dust on the Terrestrial Environment

The term "dust" is vague and covers all types of particles made by the aggregations of matter larger than individual molecules. The lower size limit of these particles is in the vicinity of 0.001 μm. The upper size limit of particulate matter is generally considered to be 100 μm, since particles larger than 100 μm fall out of the air very rapidly [Lodge et al. 1981]. In general, particle sizes can be grouped into three classes [Lodge et al. 1981]. The first class has particles below 0.1 μm. These particles generally come from combustion sources and disappear relatively rapidly due to coagulation with other particles to form larger particles. The second class are particles between 0.1 and 2 μm. These particles are generally created by the coagulation of smaller particles. Particles in this class tend to persist in the atmosphere. The third class are particles greater than 2 μm. These particles are generally the result of mechanical processes and they are readily removed by rainfall and sedimentation. The size range measured as total suspended particulates by MOEE is from 0.1 to 100 μm. Thus the very small particulates, that are short lived in the atmosphere, are not included. The total suspended particulates are, in general terms, smoke, dust, fly ash and pollen. The MOEE criteria for total suspended particulates and dustfall is given in Table D-6.4.

Table D-6.4: Summary of Ambient Air Quality Criteria for TSP and Dustfall

Pollutant	Averaging Period	Ambient Air Quality Criteria
Total Suspended Particulates (TSP)	24 hours	120 $\mu g/m^3$
	1 year	60 $\mu g/m^3$
Dustfall	1 month	7.0 g/m^2/30 days
	1 year	4.6 g/m^2/30 days

Source: IWA [1994]

Emission Sources

Landfill Sites

Six emission sources were identified from the proposed Durham landfill site.

1) Off-site traffic

2) On-site haul routes - waste haul, clay import and granular trucks, small vehicles

3) Daily landfill operations (working face) - bulldozer, landfill compactor

4) General site maintenance - grader, backhoe-loader

5) Wind erosion - stockpiles, working face, excavation, final cover, exposed soil

6) Scaper activities - loading, dumping and travel between the active areas

The dust is primarily the result of silt in the soil and gravel roads being released into the air by trucks and machinery and wind. The dust is not from the refuse itself. The dust emissions will be greatly reduced if the mitigation measures outlined in the Durham landfill report [IWA, 1994] are followed. These measures include wet sweeping and water flushing of paved roads and watering of on-site unpaved roads. Truck wheels will be cleaned to prevent off-site tracking of soil. In addition, the perimeter roads and access roads will be paved and the stockpile will be mulched and hydro-seeded to prevent wind erosion of soil. Nevertheless, there will be significant emissions of dust from the site. From the results of the short term TSP modelling of cumulative dust impacts, there were many sites that exceeded the 24 hour criteria of 120 $\mu g/m^3$ and seven receptors that exceeded 400 $\mu g/m^3$. The receptors that exceeded 400 $\mu g/m^3$ were all along the

Taunton Road. Four of these receptors were identified as having minimal change due to the proposed landfill over baseline dust levels, leaving three receptors that would exceed the 400 $\mu g/m^3$ criteria due to the operation of the landfill. Most receptors on the west, south and eastern sides of the proposed landfill site had no predicted impacts of dust. Therefore, the impact of dust from the landfill site will primarily be along the roads and these impacts are difficult if not impossible to separate from the impact due to present and future traffic along these roads. This point is supported by the fact that the highest dustfall receptor was receptor R543, which is at the corner of Brock Road and Taunton Road, over 1 km from the landfill site.

Incinerators

There are three dust emission sources at the incinerator. The first is due to truck traffic on and off the site. The second is from fugitive emissions and the third is from the stack. There was no information supplied on the dust from truck traffic or fugitive emissions. The emissions (TSP) from the stack were calculated to be 0.2 $\mu g/m^3$ over an averaging period of 24 hours. This is well below the 24 hour ambient air criteria of 120 $\mu g/m^3$. Even though the emissions from truck traffic and fugitive emissions are not known, it is expected that there will be relatively little dust from the incinerator site. This is because the access roads to the incinerator are all paved. The refuse is dumped, sorted, and loaded into the incinerator on a concrete floor within the incinerator building. The ash from the bag houses and stack is loaded directly into a tanker truck from a hopper and the bottom ash is first quenched in water and then loaded, within the building, into trucks. Nevertheless, the fly ash and bottom ash are disposed of in landfill sites. The disposal of this ash will result in the emission of dust around the landfill site, as indicated in the landfill section.

Effects on the Terrestrial Environment

The discussion of the effects of dust will concentrate primarily on plants, since animals can avoid areas with high Total Suspended Particulates (TSP) and dustfall.

A lot of research on dust effects on plants has been devoted to the toxic effects of chemically active dusts, such as cement dust, soot or salt [Linzon, 1973], rather than to the adverse effects of small particles considered to be non toxic [Eveling and Bataille, 1984]. The dust emissions from both the landfill site and the incinerator could be considered non toxic. The exact chemical composition of these dusts is unknown. In general, the composition of road dust particles greater than 2 um reflects the composition of the earth's crust and is primarily composed of iron, aluminum and silicon [Lodge et al. 1981]. The particles between 0.1 and 2 um are predominately composed of sulphate, nitrate ammonium, lead, bromine, carbon and tarry organic material [Lodge et al. 1981]. The exact composition will reflect the chemical composition of the local soil, the composition of the road surfaces as well as any soil or gravel brought into the site.

The effect of the dust from either the incinerator or landfill site on the local vegetation is difficult to predict. The main difficulty is that most research on the effects of non toxic dusts, such as road dust, on plants is either carried out in the laboratory or in the field under conditions where dust will accumulate. Since dust is readily blown and shaken from leaves as they flutter in the wind and washed from leaves during precipitation events, the effects will depend not only on the extent of dustfall but also on wind conditions and the frequency, intensity and duration of precipitation events. In addition, the interception and retention of airborne particles by plants has been shown to depend upon the size, shape and surface texture of both the particles and the plant surfaces, as well as on the ambient meteorological conditions.[Little, 1977]. Nevertheless, high dust loading can occur under dry conditions, which might lead to adverse effects on plants.

There are three possible physical effects of non toxic dust on plants [Lodge et al., 1981] The first is stomatal obstruction. The second is blocking of sunlight and the third is an increase in atmospheric turbidity to decrease the incidence of the portion of the solar spectrum responsible for photosynthesis. Stomatal obstruction is when dust particles enter the stomates of plant leaves and prevent the guard cells from closing. The result of this obstruction is increased water loss [Krajickova and Mejstrik, 1984]. As well as stomatal obstruction there may be damage to the cuticle, also leading to water loss. Eveling and Bataille [1984] report a significant increase in water loss when particles of quartz and clay were deposited on the easily wilted leaves and petals of *Coleus blumei* Benth. An increase in water loss may lead to wilting and could weaken and even kill a plant under drought conditions. However, this is unlikely to happen along the roads around either the landfill site or incinerator site, since roadside plants tend to be hardy pioneer species well adapted to drought conditions.

The blocking of sunlight by the dust layer on a leaf could lead to decreased photosynthesis. Also, it could lead to increased leaf temperatures which will affect photosynthesis. Eller [1977] reports that road dust can increase the temperature of *Rhododendron catawbiense* Michx. leaves by 2 to 4 °C, relative to clean control leaves. This is due to increased absorption of radiation above 700 nm. This increase in leaf temperature would have an effect on photosynthesis. In terms of the peaks in the action spectrum for photosynthesis, the data of Eller [1977] gives little indication that dust changed the absorptivity values of dusty compared to clean leaves in the 400 to 500 nm range or the 650 to 700 nm range. Therefore, the dust from the landfill site may increase the leaf temperature of some plants along the roadsides but is unlikely to significantly affect photosynthesis. The amount of dustfall from the incinerator is insufficient to affect leaf temperature or photosynthesis of plants in the surrounding area.

The dust from the landfill site, and to a lesser degree from the incinerator, will contribute to the turbidity of the atmosphere. The long term effect this will have on plants is unknown, but is unlikely to be significant.

In addition to the physical properties of dust that may affect vegetation, there are some chemical properties of non-toxic dust that may effect vegetation. In general, the plant species most sensitive to dust are bryophytes and lichens. This may be attributable to the lack of a cuticle in lichens and in many species of bryophytes. The dust particles may be beneficial by providing nutrients. Tamm and Troedsson [1955] found an increase of calcium, potassium, sodium and phosphorus in mosses growing near roads compared to mosses growing away from roads. Alternatively, dust particles from road dust may have a potentially detrimental effect by changing the species composition near the road. Walker and Everett [1987] found that acidophilous species, such as Sphagnum, were replaced by minerotrophic species, such as *Bryum* spp. along gravel roads in the Alaskan tundra. These effects on bryophytes and lichens are most common in the boreal forest or tundra but not in southern Ontario. In southern Ontario, road dust may have the beneficial effect of protecting leaves from air pollution. Vasiloff and Drummond [1976] report that there was a significant reduction in symptoms of ozone and sulphur dioxide injury on plants when protected by dust compared to undusted plants. Similarly, road dust can neutralize the damaging effects of acid precipitation.

Conclusion

A large component of the dust expected in the vicinity of the landfill site and incinerator is from truck traffic bringing refuse to the site. This is evident by the high dust loading along the roads to the landfill site. If the same amount of refuse is brought to both the landfill and the incinerator site, then it is assumed the dust along the roads will be equal for both operations. Additional machinery traffic will occur at the landfill site moving, compacting and covering the refuse and slightly greater truck traffic will occur at the incinerator site hauling the ash from the site.

In terms of dust, a state of the art incinerator will have less impact on the surrounding area than the landfill site. However, since an incinerator requires a landfill site for the dumping of ash, as well as material that is not suitable for incineration, such as mattresses, the dust impacts of each on the natural environment are likely to be similar and relatively minor.

D-6.5 The Effects of Noise on the Terrestrial Environment

Noise is referred to as unwanted sound. There are two component of sound, the frequency and loudness. Humans and other animals are not equally sensitive to all frequencies. Most humans cannot hear sounds with a frequency above 20,000 Hz.. However, rats can hear frequencies as high as 70,000 Hz. High frequency sounds tend to produce changes of the inner ear, whereas low frequency tones, tend to produce changes throughout the length of the cochlea [Memphis State University, 1971]. In rats, mice and guinea pigs, high frequency noise appears to be more stressful than low frequency noise [Memphis State University, 1971]. The loudness of noise is measured in decibels (db). Decibel scales are logarithmic, rather than linear. Thus, 80 db is ten thousand times as loud as 40 db. Since the frequency of sound is a factor in determining the degree of harm of a noise as well as the loudness, there are various decibel scales. The most common decibel scale is the decibel A scale, which is geared to the frequency of human hearing. On this scale a quiet library is 20 dbA, a heavy truck at 15 meters is 90 db and a jet airplane at take-off is 145 db. A sound pressure of 180 dbA can kill [Enger and Smith, 1995].

The Environmental Protection Act outlines the applicable sound level limits for landfill operations, including working face equipment and on-site vehicles are:

a) 55 dbA equivalent sound energy (Leq) during any hour of the day from 0700 to 1900 hours:

b) 45 dbA equivalent sound energy (Leq) during the evening and night time period of 1900 to 0700 hours; or

c) the ambient sound expose level due to sources of acceptable man-made activity (such as industry, commercial use or road transportation), if higher than limits in a) and b). The landfill operations are not expected to exceed these guidelines by more than 1 dbA during the operation of the landfill. The sound exposure could exceed the guidelines by up to 2 dbA when the final cover is placed on the landfill but this will be for a brief period of time.

Landfill Site

The total change in noise levels at receptors around the landfill, due to landfill operations, depended on the location of the receptor as well as the stage of operations. In general, the total change ranged from 1.3 dbA to 10.4 dbA. A total change of 10 dbA is perceived as a doubling of loudness. Under the worst case scenario, noise levels were predicted to go from a background level of 45 dbA to just over 60 dbA, representing and increase of approximately 15 dbA. This is comparable to going from the sound level in a quiet library to a the sound level of normal speech. This change was predicted to occur at the edge of an environmentally significant area at the forest edge closest to the landfill. However, this noise level was predicted to quickly drop to 47 decibels from the forest edge to the bottom of the valley slope. Therefore, noise effects on the Environmentally Sensitive Areas were expected to be relatively minor [IWA, 1994]. Species in the forest edge may be prompted to move during noisy periods but are not thought to be particulary sensitive to these noise levels.

There are several on-site noise mitigation measures incorporated into the site design. Perimeter berms would be constructed along the north and west boundaries of the site. A sound barrier wall would be constructed along the eastern site boundary. Operational berms would be constructed at the tipping face and supplementary screening berm segments will be installed adjacent to the temporary internal haul routes. In addition, most activity will take place during the day, and sound levels are expected to drop to 45 dbA at night.

Incinerator

The noise levels around the incinerator were not available at the time of writing. Nevertheless, all the unloading of refuse, sorting of the refuse, loading of the incinerator and power generation is within the incinerator building. Therefore, most of the loud noises will be contained.

Effects on the Terrestrial Environment

Noise can have a significant impact on animal life but no direct effect on plant life. Plant life may be indirectly effected through the disturbance of pollinators such as birds and bees. However, there is a paucity of information on the effects of noise from heavy equipment and trucks on the pollination or reproductive success of plants.

Most reports on the effects of noise on animals were concerned with noise levels, over 100 dbA for short periods of time [Memphis State University, 1971]. The main impact of noise may be to increase the stress on an organism. Prolonged exposure to severe stress can exhaust an organism's resources and result in death [Memphis State University, 1971]. However, the sound levels at both the landfill site and incinerator site are will below these levels and therefore, the acute effects of hearing loss or increased stress are unlikely to occur.

Nocturnal predators and nocturnal prey are likely to be most affected by high noise levels and possible hearing loss, since these animals rely on hearing to locate prey or avoid capture. Since the activity of the landfill site is greatly reduced at night and noise levels are required to drop to 45 dbA, it is unlikely the landfill site will have a large influence on nocturnal predator/ prey interactions. The greatest effect of noise levels at 60 dbA is likely to be on the breeding areas of animals sensitive to sound. Heronries are particularly sensitive to sound, but there are no heronries reported to be in the vicinity of the Durham landfill site or incinerator site.

Conclusion

In terms of noise, an incinerator, like the KMS Peel Inc.,incinerator, will have less impact on the surrounding natural environment than a landfill site like the proposed Durham site. Nevertheless, it is unlikely that either would significantly disrupt the natural environment.

D-6.6 Habitat Loss

The siting of any waste management facility will require the displacement of land from an existing use by establishment of that facility. Land uses generally fall within three main types - developed urban, agricultural or natural areas. It is not realistic to expected that any new landfill facility would be developed in an urban area due primarily to available space considerations and land values. An incinerator might be feasibility located within an industrial area. It is therefore assumed that any new facility would be placed in either an agricultural area or in a natural area.

The placement of any waste facility within an agricultural area would remove from active agricultural use including human food production an area equivalent to the active site plus access roads, parking, etc. and any area that might be subjected to contaminated emissions. Within an agricultural area, there are usually some natural areas such as woodlots, stream corridors and fencerows that would be suitable for wildllife. In natural areas, there would be a higher concentration of such habitat. The habitat is generally considered as valuable to birds, mammals, amphibians, butterflies and other forms of life, for breeding, feeding, sheltering or migration corridors.

For a landfill handling 6.6 million tonnes of waste, the actual projected area involved is 117 ha, 42.5 ha of which is actual fill area. For an incinerator site handling a total of 2.9 million tonnes, 6 ha are required, 3 ha of which is occupied by the facility, . The incinerator does require its own landfill site for the disposal of ash and non-combustible residue in addition to this area. This would be expected to be about 30 ha if the facility were to handle 6.6 million tonnes of waste. For the purposes of this excercise, the land area required for access roads and related areas is considered to be essentially equal for both the incinerator and landfill alternatives. This factor is therefore not considered as favouring either scenario.

The habitat associated with land from agricultural or natural areas is completely lost for at least the period of active (including preparation and decommisioning) operations of the waste facilities. To a high degree, revegetation of the areas following closure of the facilities can be achieved by proper site management. While such management may make the sites aesthetically acceptible as parks or golf courses, the sites will never regain their full potential as natural areas. Lands subjected to these uses are known to be depauperate in native species, to have low biodiversity and to not function as natural ecosystems. These uses tend to require active management with significant inputs of pesticide and fertilizers. The landfill sites are not suitable for construction of buildings for many years due to settling. One other consideration is that agricultural land lost to waste management is essentially lost forever from food production (ie. steep slopes prevent agricultural equipment from working); however, closed landfill sites might be acceptible for cattle pasturing if a good and suitable vegetation cover is established and the number of animals allowed to graze is not too high.

In conclusion, the amount of habitat lost to a landfill during its operation will be greater than that lost to an incinerator which handles the same volume of waste. On closure of the facility, however, the likelihood of the area being returned to some form of green space which supports vegetation and wildlife is much greater for a landfill. In most situations, these facilities are constructed in areas that are likely to experience considerable pressure from urban development in the future, and can therefore serve as important habitat for terrestrial organisms even though they are unlikely to retain the diversity of the original pre-landfill ecosystem. In addition, modern landfills are planned to include water retention areas and sometimes wetlands, which can improve the biodiversity of the area on closure.

D-6.7 Contributions to Global Warming

Operation of both landfills and refuse incinerators has relevance to the problem of global warming. Both have emissions that exacerbate the problem primarily through the release of CO_2 and methane. Both gases, together with other anthropogenic emissions, trap sunlight within the earth's atmosphere and as a consequence ambient air temperatures may increase. Dealing with the global warming phenomenon has substantial economic ramifications and requires changes in human lifestyle [Timmerman, 1988]. Failure to pay these costs now will allow global warming to proceed at a faster rate having greater overall costs and human impacts than if the situation were addressed faced immediately.

Although the scientific community is not entirely in agreement, the majority of scientists believe that climate changes will result in an average global temperature rise of 3-5 degrees with greater changes occurring at the polar regions within forty to fifty years [Canadian Climate Program Board, 1991; AES, 1994]. Precipitation patterns could shift dramatically with certain areas receiving more rain while other areas, particularly in the centre of continents would receive less. Such a situation would have major implications for crop production [Smit, 1988] and forestry [Harrington, 1987]. Storm severity including tornados and hurricanes could increase. Increased temperatures are likely to result in increases in sea levels of up to 1.5 metres. This sea level rise would have major consequences for low lying islands and coastal areas because inundation by sea water would reduce the areas available for agricultural production, human living space and natural habitat for wildlife. Natural terrestrial ecosystems would be subjected to major disruptions through additional competition from other species, stress from environmental changes, diseases and pests [Hepting, 1963], and loss of suitable habitat [Schneider, 1989]. A major limiting factor for natural ecosystems is the capability or range extension rate for species relative to the rate of environmental change. For example, under some models, sugar maple would die out from its current Ontario range [Roberts, 1989].

A summary of implications of global warming for Canada [Jackson, 1992] suggests that the boreal forest west of Quebec may be eliminated, primarily by forest fire. Such a trend had been established during the 1980's, was halted during the cooler and wetter years of 1992-1994 in response to dust emissions from volcanoes such as Mount Pinatubo. The large number of forest fires across northern Canada in 1995 suggests that the trend has resumed.

Global warming may have benefits to Canadian production of wheat and grain corn relative to other countries as these crops may be grown further north than presently occurs [Smit, 1989]. The production of these crops would be highly vulnerable to drought as the projected climate would be outweighed by increased evaporation [Jackson, 1992]. Production of barley and oats may diminish. With respect to Ontario agriculture, reduced rainfall would render corn and soybeans vulnerable to drought [Smit, 1987]. These crops may have new potential for production in Northern Ontario although the soils may not be suitable because of texture and depth limitations. An overall assessment is that global warming would cost Ontario agriculture $170 million per year in lost production.

In response to the need for reductions in greenhouse gas emissions, 167 countries including Canada signed the United Nations Framework Convention on Climate Change in 1992. Having signed the convention, it was necessary to determine how to meet the commitments stabilization at 1990 levels by 2000 [Mortimer, 1993]. Such commitments require the involvement of Ontario. To meet this, potential instruments include regulations with respect to performance standards and building codes, proactive policies involving research and public transportation, and energy taxes. All jurisdictions in Canada are considering further actions to limit greenhouse gas emissions. A strategy for Ontario was proposed in a document prepared by the Ontario Ministry of Energy [Ministry of Energy, 1989].

In 1985, anthropogenic CO_2 global emissions were 5 billion tonnes of which Canada contributed 102 million tonnes (2%) [Hengeveld, H.G. 1988]. With 0.5% of the world's population, this is about four times the average global inhabitant output.

Table D-6.5: Gases creating greenhouse effect [Riggs, 1990]

Gas	Percent of Emissions	GWP
Carbon Dioxide	49 %	1
Methane	18 %	11
Chlorfluorocarbons	14 %	≤7100
Nitrous Oxide	6 %	270
Tropospheric Ozone	13 %	-
GWP is relative ability of indicated gas to create greenhouse effect relative to an equivalent amount of carbon dioxide [Hengeveld, 1992]		

D-6.7.1 Biological Effects

Work by Cannell and Smith [1986] suggested that a moderate climate warming of 2°C would increase the incidence of warm springs. The effect would be to advance flowering of apple trees by 18 to 24 days and advance bud break of Sitka spruce by 5 days. This would render the apples more susceptible to frost damage.

The dry weight of Virginia pine (*Pinus virginiana* Mill.) responded to increased CO_2 concentrations of 340 to 940 μl/l by increasing from 4.4 to 11.0 g/plant (150% increase) [Luxmore et al. 1986]. The increased growth was accompanied by significant increases in uptake of nitrogen, calcium, aluminum, iron, zinc and strontium. At higher rates of CO_2 (1400 μL/L) under high light intensity, Ehret et al. [1985] noted chlorosis injuries to bean (*Phaseolus vulgaris* l. cv. Pure Gold Wax).

Leaves which exhibited injury accumulated more starch than uninjured leaves and showed reduced rates of photosynthesis. Data from soybean experiments inserted into a mathematical model predict increases of 32% seed yield when the CO_2 concentrations rise from 315 to 630 ppm [Allen et al, 1987]. Neither photosynthesis nor seed yield increase in direct proportion to the ambient CO_2 level.

Under conditions of higher concentrations of CO_2, the density of stomates on leaves of several plant species decreased [Woodward, 1987]. The effect was apparently related to rate of expansion of leaves. This in turn was accompanied by greater efficiency of water use by the plants.

A study of an old field plant community showed that the effects of atmospheric enrichment by CO_2 [Wrayet al 1987] could alter the relative composition of two dominant plant species. Increasing CO_2 from 350 to 500 or 650 μl/l increased the growth rate of *Aster pilosus* Willd. Under normal circumstances, the aster is replaced by broomsedge (*Andropogon virginicus* L); however, under enrichment by CO_2, the aster was able to outcompete the grass. This study indicates that elevated atmospheric CO_2 can affect ecological processes.

D-6.7.2 Greenhouse Gas Emissions from Incinerators and Landfills

Combined emissions from the model landfill and flare over a period of 40 years amounts to 185,000 tonnes of methane and 1,756,000 tonnes of CO_2. Using the GWP equivalent of 11 for methane, this represents the same greenhouse gas emission as 3,788,000 tonnes of CO_2. In the case of the incinerator, calculations can be made from the 1993 - 1994 emissions data of the KMS Peel Inc.,facility to determine total CO_2 emissions on a per tonne of MSW basis. Assuming that a facility at the site handled 6.6 million tonnes of MSW, the total emissions were determined to be 6.73 million tonnes of CO_2. This is about 77% greater that the CO_2 equivalent from the landfill for the same MSW volume.

Calculation of CO_2 Emissions from Incinerator

Stack gas emission rate = 30 m^3/s
= 108,000 m^3/hr
@ 8.0 % CO_2 = 8640 $m^3 CO_2$/hr
@ 1.976 kg/m^3 this is 17.073 tonnes/hr
= 409.7 tonnes/day
@ 400 tonnes MSW/day
= 1.02 tonnes CO_2/tonne MSW

Table D-6.6: Comparison of greenhouse gas emissions from operation of a landfill and incinerator of equal size (handling 6.6 million tonnes) (Values reported in '000 tonnes)

Gas	Incinerator		Landfill	
	Raw[1]	GWP[2]	Raw	GWP
CO_2	6730	6730	1756	1756
Methane	NA	0	184	2032
Total		6730		3788

Raw is basic tonnage of gas indicated,
GWP is calculated equivalent CO_2 amount using 11 x factor for methane

Recent information indicates that the GWP of 11 for methane may be too low. In the "Canada's National Action Program on Climate Change - 1995" a value of 24.5 is suggested. If this value is used, the contribution of the landfill increases to 6.288 million tonnes, and results in a basic equivalency of the two methods with respect to these two greenhouse gases. A recent Ontario report on the status of greenhouse gas emission reductions [MOEE, 1995] uses a GWP factor of 21 for methane.

The above assessment assumes that there is a landfill gas collection system in place at the landfill which is functioning at 50% collection efficiency from the fifth year of operation until the 20th year after opening and 70% efficiency for the next 20 years. Flaring efficiency is assumed to be 98%. Landfills of a similar size without gas collection systems would produce a global warming potential considerably larger than the above projections.

As noted in Section D-6.1.5, N_2O is not thought to be a significant contributor to global warming. Although N_2O has a very high GWP, it is not produced in significant quantities by state-of-the-art incinerators or landfills.

D-6.8 Pest Species

In the past, landfill sites have been justifiably considered as reservoirs of pests and undesirable animal species. Providing that new landfill sites are diligently operated to modern management standards, the pest problems experienced with old sites should be kept to a minimum.

Birds

Sanitary landfill sites are well-known as being attractive to birds generally regarded as nuisance birds. These include primarily gulls (Ring-billed Gull and Herring Gull), European Starling, American Crow and Common Raven. The latter is common on sites on the Precambrian Shield in Ontario but is gradually re-occupying its former range further south. Other bird species may occasionally use landfill sites.

Large numbers of these nuisance species regularly concentrate to feed at the operating face of landfill sites across Ontario according to Christmas Bird Census. Counts of 2000 gulls, 200 crows and 200 starlings are not unusual. Predicted daily maximum numbers for the proposed Durham landfill were 5000 and 400 starlings and crows respectively. The presence of the birds at the landfills in itself is not a major concern; however, there is a concern when birds are moving between roosting sites and the landfill, and their route crosses the flight path of aircraft. The choice of landfill sites should bear this significant hazard in mind. This problem was particularly acute at North Bay where the landfill was not far from the airport runway and consequently had to be relocated. The birds do deposit excrement which is unsightly and a possible small health risk but this is minor in relation to the activity of the landfill itself. On the other hand, some birds are likely to become a nuisance at sites near a landfill simply because of their tendency to concentrate in the vicinity of a landfill. The nuisance effects include noise, redistribution of garbage, and damage to crops by feeding on the seeds and seedlings at the beginning of the growing season. Damage to crops may also occur prior to harvest by feeding on the grain or by exposing the grain to disease organisms (ie. opening corn husks to attack by the *Fusarium* fungus which produces mycotoxins rendering the grain unsuitable for food.

The attraction of landfills to birds has a side benefit in that less common species visit such sites to the delight of bird watchers. Examples of these include Glaucus, Iceland and Black-backed Gulls.

Mammals

The principal rodents of concern at landfills are Norway rats and mice which transmit or at least harbour agents of disease dispersal (ie fleas). Other scavengers at landfills include raccoons, opossums (which are spreading in Ontario), skunks and bears (not in southern Ontario). Proper site management should control this problem to a large extent by daily covering of the putrescible materials upon which these species feed. Fencing may exclude these species in part; however, access through open gates and by delivery within loads of garbage is still possible.

Insects

Decomposing organic matter is attractive to insects although proper site management (daily covering of garbage) should significantly curb insect reproduction by controlling access to such matter. In particular, house flies are potential problems as they may disperse disease organisms in the vicinity of the landfill resulting in food contamination for humans, for example. Other types of flies, such as fruit flies, may be more of a problem near a landfill by feeding on and damaging fruit. Lastly, mosquitos may find good breeding sites at landfills if water-holding vessels are not properly covered. This concern may extend to wetlands that may be constructed when the sites are closed or if water is allowed to pool. This problem may be lessened if the water depth is adequate (mosquitos generally prefer shallow pools). Currently in Ontario, transmission of human diseases by mosquitos is not regarded as a major concern. However, this may become more significant over time, especially if global warming occurs as predicted and species of mosquitos which transmit diseases such as encephalitis are then able to survive in Ontario.

Weeds

Improperly maintained landfill sites are potential sources of weed seeds. In particular, wind-disseminated seeds of species such as dandelion, thistles, sow thistle, or milkweed may compete with agricultural crops. These weeds may also harbour insects and disease organisms that could be spread to infest the crops. In properly managed sites (ie. weeds are cut before producing viable seed), this should not be a problem.

D-6.9 Conclusions

The risk of direct adverse effects to the terrestrial environment from chemicals emitted by properly functioning state-of-the-art incinerators or landfills, operating under the design assumptions used in this assessment, is anticipated to be insignificant to populations of terrestrial receptor species.

The largest differences in impact of the two waste management methods on the environment appear to be through the amount of land utilized, and thus ecologically altered, and through the quantities of nitrogen oxide emissions. Landfills directly occupy considerably more space than do incinerators, and hence more land is altered. However, most of this land can be returned to greenspace upon closure of the facility. Although the quality and diversity of the resultant ecosystem is often reduced compared to its original state, the site can provide important habitat for wildlife as well as valuable greenspace for urban residents, if properly managed.

Incinerators emit considerably more nitrogen oxides than do landfills. A large-scale switch from landfills to incinerators could result in a significant increase in nitrogen oxide emissions. Since there is some evidence that upland hardwood forests in some parts of Ontario may already be under stress from excess nitrogen oxide loadings, such a switch could hasten the development of suspected impacts on forests in sensitive areas. Such adverse effects must be balanced against the benefits to nitrogen loving plants, such as many agricultural crops. It should be noted that total emissions from incinerators would be a small fraction of total emissions in Ontario.

The concentrations of mercury in tissue in hawks and woodcock that have been predicted by the modelling of incinerator emissions exceed concentrations at which effects have been noted in some sensitive species of birds. A number of factors in the exposure modelling have resulted in a high degree of conservatism in the estimates; hence, the actual exposures are likely to be lower than projected. A more detailed assessment for mercury, with more sophisticated modelling, would be required to characterize the risk to raptors and to birds that consume soil invertebrates. Mercury emissions to the air from landfills were not assessed due to a lack of data on emission rates.

This assessment indicates that concentrations of carcinogenic volatile organic compounds (VOCs) at the maximum point of impingement near a landfill are below levels that would be anticipated to have meaningful effects on populations (set at a risk of 1:100). However, they are only slightly lower than these levels and could be regarded as posing a potential for concern given the uncertainty in the designation of the acceptable risk level.

There is very little information available on the toxicity of some of the compounds to terrestrial species, notably 2-Methyl-fluorene, the mercaptans and Bromodichloromethane; hence the conclusions regarding these compounds are subject to additional uncertainty. Some literature indicates that some of the chlorinated VOCs may be highly toxic to bird embryos. However, there was not enough information available on transfer from adult bird tissue to the embryo or on the effect of these concentrations to determine if the concentrations present near a landfill could produce an effect on embryos. Estimated concentrations are clearly far too low to affect adult or young birds.

Landfills produce far less carbon dioxide than do incinerators; however, they produce more methane. Since methane has an effect as a greenhouse gas of many times that of CO_2, the overall difference between the two disposal options in the production of greenhouse gases is highly dependent upon the equivalency factor used. Although a factor of 11 was used in the calculations for this assessment, as has been used in literature over the past five years, some recent documents have used factors from 21 to 24.5. There is some uncertainty as to the size of this factor; hence, the two disposal methods must be viewed as being roughly equivalent within current levels of uncertainty. This assumes that the landfill has a fully functioning gas collection system operating at 50% collection efficiency from the fifth to the 20th year after opening and at 70% for the next 20 years. Landfills without gas collection are likely to produce more global warming potential than comparable volume state-of-the-art incinerators, whereas landfills with significantly more efficient gas collection systems could produce less. There would appear to be significant potential for reduction of greenhouse gas emissions through improving the management of methane emissions at landfill sites. A complete comparative analysis of greenhouse gas emissions is beyond the scope of this study; however, it would be useful in the assessment of waste disposal alternatives.

In a review of assessment surveys and complaint investigations conducted by the Phytotoxicology Section around landfills and incinerators over the past 20 years, no evidence was found of adverse terrestrial impacts around properly constructed and properly operating facilities. Adverse impacts, most of which were correctible, did occur in some cases as a result of design faults (including lack of gas collection and venting systems), failures in systems, or poor drainage measures. The conclusion is that proper design, operation and maintainance of waste management facilities is important for the prevention of adverse ecological effects.

Additional Modelling for Mercury from Incinerator Meeting A-7 Guidelines

Since the concentrations of mercury in bird tissues predicted in a worst case scenario by the modelling process for areas near existing state-of-the-art incinerators were found to exceed the lowest levels at which effects have been observed, it was decided to run the hawk exposure model for an incinerator that meets the existing Ontario A-7 guidelines. Since the hawk was the most sensitive of the chosen receptors, this would determine whether there is potential for effects resulting from such an incinerator. An incinerator with the same dispersion characteristics as that previously modelled, but meeting the A-7 criteria would result in a maximum 24 hour ground level concentration of 0.0029 μg Hg/m^3, in contrast to 0.0215 μg Hg/m^3 for the existing facility, as previously modelled. Exposure modelling using the CAPCOA model results in a maximum addition to the concentration of mercury in the hawk tissues of 0.905 μg/g, as compared to 6.7 μg/g being added by the existing facility. Since the additional mercury resulting from an incinerator meeting the A-7 guidelines is considerably below the lowest of observed effects concentrations of about 4.3 μg/g in bird tissue, the effects of such an incinerator would be expected to be negligable where background soil mercury levels are low. Since the exposure scenario used in the modelling is a worst case situation in many respects, the additions of mercury due to the presence of an incinerator meeting the A-7 guidelines are expected to be minor in relationship to background concentrations.

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APPENDIX D-I

SYNOPSIS OF PHYTOTOXICOLOGY SECTION FILES RELATING TO LANDFILLS AND INCINERATORS

Synopsis of Phytotoxicology Section Files Relating to Landfills and Incinerators.

1987
ARB-109-87-Phyto
Bastard Twp
Portland
Phytotoxicology complaint investigation on the xxxx property, Portland, Ontario - August, 1986.
Alledged effects of burning garbage on nearby crops, i.e. concerns regarding contamination of crops and suitability as food. Scan of 22 elements in soil and grain crop were all normal except for elevated Mn in the grain but this was not expected to consitutae a problem. No adverse effects were observed.

1971
None
Memo
Berrill-Trustrum Landfill
Mississauga
Histological investigation - Samples collected on vegetation complaint investigation, Mrs. xxxx, Mississauga on November 10, 1971.
Growth increments of trees and assessment of roots. Trees were dead; growth decline began when landfill started to operate; mycorrhizae diminished. Effects attributed to oxygen depletion due to presence of gas (methane) in soil in high concentrations.

1971
None
Memo
Berrill-Trustrum Landfill
Mississauga
Repeat visit to the property of Mrs. xxxx (same as above), Mississauga, on Novemebr 10, 1971.
Methane gas causing tree death
Progress report on tree roots and foliage, soil subjected to analysis for 11 chemical elements

1972
None
Memo
Berrill-Trustrum Landfill
Mississauga
Bioassay results of soil samples collected from the property of Mrs xxxx November 10, 1971
Bioassay of soil for phytotoxicity
No negative effect of soil on seed germination & growth but soils could have degassed

1971
None
Memo
Berrill-Trustrum Landfill
Mississauga
Vegetation complaint investigation to the property of Mrs. xxxx, Mississauga, on June 22, 1971.
Vegetation injury alledged and associated with foul soil odour from landfill
Confirmed significant injury to pine, spruce and cedar. Attributed to oxygen depletion due to presence of gas (methane) n soil in high concentrations.

971
None
Memo
Berrill-Trustrum Landfill
Mississauga

Repeat visit to the property of Mrs. xxxx (same as above), Mississauga, on September 16, 1971.
Vegetation injury alledged
Confirmed significant injury to trees continuing to progress.

1973
None
Memo
Berrill-Trustrum Landfill
Mississauga
Berrill-Trustrum Land-fill Site
Assessment of efforts to reduce methane problems associated with the landfill
Soil flora and fauna returning to normal after abatement measures installed

1976
None
Letter
Berrill-Trustrum Landfill
Mississauga
No title
Alledged death of cedar hedge (complaint)
Pathological assessment of soil and cedar indicated transplant problems

1992
ARB-019-92-Phyto
Berrill-Trustrum Landfill
Mississauga
Phytotoxicology complaint investigation on the xxxx property, Mississauga - October 31, 1991.
Alledged contamination of property due to proximity to landfill leading to court case
Scan of 18 chemical elements indicated normal concentrations in soil. No mention of any observed effects.

1992
ARB-259-92
TM
Biggar's Lane Landfill
Brantford
Biggar's Lane Landfill, Township of Brantford
Trees planted adjacent to landfill are dying
Visual inspection indicated transplant shock and drought as cause of mortality

1980
None
Black Creek Landfill Site
Abandoned landfill site inspections, Borough of York - June, 1979
Closed prior to 1960
Visual only, no injury, greenbelt

1979
None
Letter
Brittania Road Landfill
Mississauga
No title
Alledged dieback of poplar trees but landfill not yet open
Pathological diagnosis of Dothochiza canker disease

1987
ARB-115-087-AQM
Brock West Landfill
Pickering
Air Quality Survey in the vicinity of the Brock West Landfill Area, Pickering, Ontario, March & April, 1987
Air quality indicated odour problems
141 organics and reduced sulphur compounds in low concentrations

1987
ARB-115-87
Brock West Landfill
Pickering
Phytotoxicology investigation in the vicinity of the Brock West Landfill Site, Pickering - 1986
Request for Phytotoxicology survey by Minister
Scan of 22 elements in birch indicated only elevated Cl at 1 of 6 sites. Dendrochronology indicated reduced tree growth at only one of six sites. No visible injury to vegetation except possible drainage problem, or effects from recreational uses near the affected tree.

1980
None
Coronation Park
Abandoned landfill site inspections, Borough of York - June, 1979
Closed 1940's
Visual only, no injury, parkland with tennis courts

1984
ARB-179-84-Phyto
East Whitby Landfill
East Whitby
Examination of dead vegetation at sanitary landfill site, Township of East Whitby, August 15, 1984.
Leachate from landfill with GM wastes alleged to have killed trees
Suite of 18 chemicals, only B and Cl elevated, tree mortality likely due to high water table

1987
ARB-118-87-Phyto
Essex County Landfill
Maidstone Twp
Phytotoxicology assessment survey investigation in the vicinity of the Essex County Landfill, Maidstone Township - November, 1986. No increases in metals or PCBs near landfill. No mention of any observed effects.

1989
ARB-238-89-Phyto
Glenridge Landfill Site
St Catharines
Phytotoxicology assessment survey investigation in the vicinity of the Glenridge Landfill site, St. Catharines - 1989.
Alledged vegetation injury
Visible injury to vegetation was associated with liquid flowing from drainage pipe from landfill. An old storm sewer collection pipe within the old quarry was not connected to the the sewer system as was the rest of the leachate collection system, but discharged from a pipe partway down the escarpment..

990
RB-130-90-Phyto
omplaint
lenridge Landfill Site

St Catharines
Phytotoxicology complaint investigation on the xxxx property, St Catharines - August 17, 1989 and July 12, 1990.
Landfill leachate problem. Eruption of sewer line carrying leachate.

1990
ARB-132-90-Phyto
Complaint
Glenridge Landfill Site
St Catharines
Phytotoxicology complaint investigation on the xxxx property, Ave., St Catharines - August 16, 1989 and July 12, 1990.
Landfill leachate problem. Eruption of sewer line carrying leachate.

1990
ARB-159-90-Phyto
Complaint
Glenridge Landfill Site
St Catharines
Phytotoxicology complaint investigation on the xxxx property, St. Catharines - August 17, 1989 and July 12, 1990.
Landfill leachate problem. Eruption of sewer line carrying leachate.

1990
ARB-160-90-Phyto
Complaint
Glenridge Landfill Site
St Catharines
Phytotoxicology complaint investigation on the xxxx property, St. Catharines - August 17, 1989.
Landfill leachate problem. Liquids from storm drainage pipe in landfill discharging upstream.

1990
ARB-201-91-Phyto
Complaint
Glenridge Landfill Site
St Catharines
Phytotoxicology complaint investigation on the xxxx and xxxx properties, St. Catharines - 1991.
Landfill leachate problem

1980
None
Jane-Bloor Landfill Site
Abandoned landfill site inspections, Borough of York - June, 1979
Closed in late 1930s
Visual only, no injury, theatre and parking lot

1991
ARB-166-91
Lambton County Landfill
Sarnia
Phytotoxicology assessment survey in the vicinty of Lambton County Landfill Site, Sarnia-1991
Emissions from landfill affecting nearby soils. Trees in adjacent woodlot dying.
Scan of 9 metals, all below ULN, no PCB's. Cause attributable to poor drainage resulting from construction of berms, sedimentation ponds and drainage works at the landfill. No evidence of injury due to chemical toxicity.

1980
None
Lapp-Avon Landfill Site

Abandoned landfill site inspections, Borough of York - June, 1979
Closed in 1930s
Visual only, no injury, residential area

1985
None
Markdale Historic Landfill
Markdale
None available
Tree deaths along river possibly related to old landfill
Visual only, flooding by beaver activity believed to be cause of tree mortality

1992
ARB-264-92
TM
Mountain Road Landfill
Niagara Falls
Assessment of vegetation - Mountain Road Landfill Site, - Niagara Falls (1992).
Assessment requested in view of proposed expansion to landfill
Visual only, water ponding and excess soil over tree roots caused death of some trees.

1980
None
Outlook Landfill Site
Abandoned landfill site inspections, Borough of York - June, 1979
Closed prior to 1940
Visual only, no injury, senior citizen apartments and parking

1985
ARB-174-85-Phyto
Park Road Landfill
Grimsby
Phytotoxicology investigation on the xxxx property next ot the Park Road Landfill site, Grimsby - July 9, 1985.
Alledged contamination of field and woodlot by leachate from landfill. No injury was found that could be attributed to landfill leachate. Scan of 17 elements in soil and foliage showed only normal levels

1972
None
Pedersen Disposal SIte
Uxbridge
Investigation of a vegetation injury complaint in the vicinity of the Perersen Disposal Site, Uxbridge, on June 28, 1972
Alledged vegetation injury from open burning (complaint).
Scan of 5 chemical elements indicated slightly elevated chloride in soil. Death of trees was attributable to flooding resulting from drainage changes due to two roads and a railway. Smoke or fume emissions were ruled out as a cause of injury, with the exception of one small area where direct fire damage (scorching) was observed.

1993
023-3511-93
TM
Pelham Road Landfill
St. Catharines
Phytotoxicology survey report: Pelham Road Landfill Site - St. Catharines (1992)
Proposal to make public park at site, methane causing bubbling mud.
18 elements at below ULN, 20 organics in soil and water below detection, trace PCB, some vegetation injury possibly due to water ponding.

1980
None
Porter landfill Site
Abandoned landfill site inspections, Borough of York - June, 1979
Closed prior to 1960s
Visual only, no injury, football field/stadium

1982
None
Ridge Landfill
Blenheim
Vegetation and soil assessment survey in the vicinity of the Ridge Landfill site, Blenheim - September, 1980.
Potential effects of emissions from liquid waste solidification facility
16 inorganic elements in soil, crop (soybean, corn) and silver maple foliage and twigs were normal but P, K, Cd, Mn and B were slightly elevated at distant sites. No effects of landfill noted. Concluded that current and historic use of the landfill site has not resulted in any related off-property chemical contamination or foliar injury.

1982
ARB-268-92
TM
Storington Landfill
Kingston
Results of Phytotoxicology investigation Re: Decline of trees at north limit of Storington Landfill site: July 12 1992.
Death of trees noted near edge of landfill proposed for re-opening.
Visual inspection only. Trees mechanically injured or affect by excees soil on roots.

1992
ARB-049-91
Truscan Property
Vanier
Phytotoxicology Section investigation in the vicinity of the Truscan property, Landry Street, Vanier on August 9 and November 28, 1990.
Preliminary indications of elevated metals at historical (1930s) landfill site. Subdivision built on site.
Scan of 30 elements indicated soil ULN and/or Clean-up Guidelines for As, Cu, Pb, Hg, Mo and Se were exceeded. SB, Fe and Zn exceeded ULN. Na, Se and Mo exceeded vegetation ULN

1992
ARB-003-92
Unnamed landfill at Dixie Rd and Hwy 5
Mississauga
Phytotoxicology complaint investigation on the xxxx property, Mississauga, August, 1991.
Grass dying on alledged old landfill site, open burning of grass close to methane vent
Determined that injury was due to white grubs and chinch bugs

1983
ARB-057-83-Phyto
Upper Ottawa Street Landfill
Hamilton
Phytotoxicology survey in the vicinity of the Upper Ottawa Street Landfill site. Hamilton, 1981. Traces of organics including PCB, and pesticides found in leachates, organics including freon, tri & tetrachloroethane
Scan of 14 elements in soil, maple foliage, moss bags and vegetables were all considered normal except for one site where soil As was elevated but not related to the landfill. Some evidence of As in moss bags decreasing with increasing distance from landfill, but concentrations were quite low.

1980

None
Wilby Landfill Site
Abandoned landfill site inspections, Borough of York - June, 1979
Closed early 1950's
Visual only, no injury, parking lot and ravine

1975
None
Commissioners Street Incinerator
Toronto
Phytotoxicology heavy metal survey - 1976, Commissioner Street Incinerator - Toronto
Potential impact of emissions of incinerator
Scan of 5 chemical elements, historical impact of emissions with Pb, Cd, and Zn accumulating in soil near facility.

1989
ARB-148-88-Phyto
GC
Highland Creek Sewage Treatment Plant
Phytotoxicology Soil Survey in the vicinity of the Highland Creek Sewage Treatment Plant Incinerator: Scarborough - 1987.
Public concerns about emissions from incineration of sludgeScan of 16 inorganic elements showed normal levels, no PCBs detected, Dioxins and furans detected not related to incinerator.

1990
ARB-263-89-Phyto
GC
Highland Creek Sewage Treatment Plant
Supplementary Soil Survey in the vicinity of the Highland Creek Sewage Treatment Plant Incinerator: Scarborough - 1989
Public concerns about emissions from incineration of sludgeEssentially same results as in 1987 study. Indication of on-site accumulation of Cu, Cd, and Cr. probably resulting from fugitive emissions from ash disposal.

1985
ARB-013-85
SWARU
Hamilton
Concentrations of PCDD and PCDF in soil from the vicinity of the SWARU incinerator, Hamilton
Dioxins emmitted from incinerator
Dioxins in soil mainly non-detectable except dominated by O8CDD (max 3500 pg/g) and H7CDD (max 390 pg/g). Furans variable with peak of 180 pg/g H7CDF. Dioxins not accumulating in surface soils as a result of the incinerator.

1994
SDB-038-3512-94
SWARU
Hamilton
Dioxins in tree foliage collected near SWARU, Hamilton, 1993
Dioxins emmitted from incinerator
Dioxins and furans in cedar foliage ranged from 0 pg/g to 37 pg/g T4CDD and 70 pg/g T4CDF close to SWARU in a clear gradient

1986
ARB-125-86-Phyto
GC
Victoria Hospital
London
Pre-operational survey in the Westminster Ponds/Pond Mills Conservation Area relative to the Victoria Hospital Energy from Waste (EFW) power plant, London, Ontario, 1984.

Pre-operational study
Scan of 19 chemical elements in soil, vegetation and mossbags not related to incinerator

1992
ARB-053-92-Phyto
GC
Victoria Hospital
London
Phytotoxicology surveys in the vicinity of the Victoria Hospital Energy from Waste (EFW) power plant, London, Ontario, 1984.
Incinerator in operation for 3 years
Scan of 22 elements in soil, vegetation and moss. Scattered exceedence of ULN unrelated to incinerator.

1993
011-3511-93
Victoria Hospital
London
Phytotoxicology surveys in the vicinity of the Victoria Hospital Energy from Waste (EFW) power plant, London, Ontario, 1991 and 1992.
Incinerator in operation for 5 years
Scan of 18 inorganic elements showed mainly normal levels except possibly Zn and Mn.

Summary of Investigations in the Vicinity of Landfills

Phytotoxicology Section has conducted nearly 40 investigations around 26 landfills. Injury was observed in 13 cases (not including the multiple cases of the same problems at Berrill-Trustam and Glenridge). In six of these cases, injury was attributable to the landfill. Four of these resulted from surface drainage problems and ponded water. One (8 investigations at Berrill-Trustam) resulted from high concentrations of methane in the soil produced from a facility with no methane collection, venting or flaring systems. The other (6 investigations at Glenridge) resulted from the failure of an improperly designed or built leachate collection system. At three sites slightly elevated levels of some contaminants were detected (Cl at all three, B at one and As at another).

Summary of Investigations in the Vicinity of Incinerators

Phytotoxicology Section has conducted eight investigations in the vicinity of incinerators (including MSW and sewage sludge). In no cases were injury or adverse effects observed. There was some evidence of the accumulation of contaminants in the vicinity of three of the facilities, including Pb, Cd, and Zn in soils probably resulting from the handling of ash at Commissioner St., dioxins in cedar (at SWARU) and possibly Zn and Mn in vegetation near Victoria Hospital.

APPENDIX D-2

CAPCOA MODEL OUTPUTS

Incinerator-vegetation model

CHEMICAL	Ground Level Concentration GLC (data input) (ug/m3)	Deposition to Vegetation DEP = (GLC*Deprate*sec/d) (ug/m2/d)	Henry's Law Constant H (data input) (Torr.-L/mol-kelvin)	Weathering Constant Non-volatiles - K = 0.0693 (1/d) Volatiles - Rv = Vda/Mf*H/RTe*1000/(0.9+(0.1*Kow))
Benzene	1.30E-01	3.93E+01	4.16E+03	2.10E+03
Dioxin/furans	1.38E-08	2.38E-06	n.a.	6.93E-02
Mercury	2.15E-02	3.72E+00	n.a.	6.93E-02
2-methyl fluorene	7.14E-04	1.23E-01	n.a.	6.93E-02

CHEMICAL	Deposition concentration vegetation Cdepv = [DEP*IF/(K*Y)]*[1-EXP(-KT)] (ug/kg)	Bioavailability Factor BIO (data input)	Soil half-life t1/2 (data input)* (d)	Soil Elimination Constant Ks = 0.693/t1/2 (1/d)	Integral function X = [(EXP(-Ks*Tf)-EXP(-Ks*To))/Ks]+Tt
Benzene	3.66E-03	1.00E+00	1.60E+01	4.33E-02	7.28E+03
Dioxin/furans	6.70E-06	4.30E-01	4.38E+03	1.58E-04	2.97E+03
Mercury	1.04E+01	1.00E+00	4.90E+03	1.41E-04	2.75E+03
2-methyl fluorene	3.46E-01	1.00E+00	4.80E+02	1.44E-03	6.61E+03

CHEMICAL	Soil conc. Cs = (DEP*X/(Ks*SD*BD*Tt) (ug/kg)	Octanol:water Kow (data input)**	Carbon part. coeff. Koc (data input)**	Uptake Factor UF2 = [(0.03*$Kow^{0.77}$)+0.82]/[(Koc)(Foc)]	Translocation conc. vegetation Ctrans = (Cs*UF2)*** (ug/kg)
Benzene	1.36E+01	1.52E+02	8.90E+01	2.53E-01	0.00E+00
Dioxin/furans	9.20E-05	8.00E+06	3.30E+06	1.88E-02	1.73E-06
Mercury	1.48E+02	N.A.	N.A.	9.00E-02	1.34E+01
2-methyl fluorene	1.16E+00	1.51E+04	7.30E+03	6.91E-02	8.02E-02

CHEMICAL	Final vegetation concentration Cfv = (Cdepv*BIO)+Ctrans (ug/kg)	Final vegetation concentration (ppm) conversion = Cfv/1000 (ug/g)
Benzene	3.66E-03	3.66E-06
Dioxin/furans	4.61E-06	4.61E-09
Mercury	2.38E+01	2.38E-02
2-methyl fluorene	4.27E-01	4.27E-04

Incinerator-vegetation model

GLC = 24 Hr. average maximum ground level concentration utilized
Deprate = deposition rate of 0.0035 m/s was utilized for volatile chemicals
= deposition rate of 0.002 m/s was utilized for chemicals other than volatiles
sec/d = number of seconds per day = 86,400
IF = interception fraction for vegetation (grass and leafy plants) = 0.39
K = weathering constant for non-volatile contaminants = 0.693/10 (1/d)
Rv = weathering constant for volatile contaminants (1/d)
Vda = 302 m/d
Mf = vegetation yield = 2.0 kg/m2
H = Henry' Law Constant (Torr.-L/mol-kelvin) = (Pa m2/mol x 7.5)
R = Gas Constant = 62.4 Torr.-L/mol-kelvin
Te = temperature = 298°K
Kow = octanol:water constant
Y = seasonal vegetation yield (grass and leafy plants) = 2 kg/m2
EXP = exp base e = 2.72
T = growth period for vegetation (grass and leafy plants) = 90 days
* Soil half-life for mercury based on 5% annual soil loss to volatilization and leaching.
* Soil half-life for 2-methyl fluorene based on CAPCOA value for PAH's.
To = beginning of evaluation period = 0 days
Tf = end of evaluation period = 7300 days (20 year life expectancy of facility)
Tt = total exposure period (Tf-To) = 7300 days (20 year life expectancy for facility).
SD = soil mixing depth = 0.05 m
BD = soil bulk density = 1,333 kg/m3
** Kow and Koc value for fluorene used for 2-methyl fluorene
For UF2, 0.03, 0.77 and 0.82 are empirical constants
*** for volatiles, the uptake of volatiles from soil through roots is assumed to be zero.
Foc = fraction organic carbon in soil = 0.1%

Incinerator-cattle model

CHEMICAL	Ground Level Conc. GLC (data input) (ug/m3)	Water concentration Cw (data input) (ug/kg)	Water Ingest Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vegetation conc. Cfv (data input) (ug/kg)	Final Feed conc. Cff (data input) (ug/kg)	Vegetation Ingestion Dose VID = (Cfv*%G*Flg)/BW (ug/kg/d)
Benzene	1.30E-01		0.00E+00	3.66E-03	3.66E-03	7.84E-05
Dioxins	1.38E-08		0.00E+00	4.61E-06	4.61E-06	9.88E-08
Mercury	2.15E-02		0.00E+00	2.38E+01	2.38E+01	5.10E-01
2-methyl fluorene	7.14E-04		0.00E+00	4.27E-01	4.27E-01	9.15E-03

CHEMICAL	Feed Ingestion Dose FID = [(1-%G)*Flf*%F*Cff]/BW (ug/kg/d)	Soil Ingestion Rate SI = [(1-%G)*%Sf*Flf]+%G*%Sp*Flg (kg/d)	Soil concentration Cs (data input) (ug/kg)	Soil Ingestion Dose SID = (Cs*SI)/BW (ug/kg/d)	Total Ingestion Dose TID = (WID+VID+FID+SID) (ug/kg/d)	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)
Benzene	6.37E-06	5.62E-01	1.36E+01	1.53E-02	1.54E-02	2.08E-02
Dioxins	8.03E-09	5.62E-01	9.20E-05	1.04E-07	2.10E-07	2.21E-09
Mercury	4.14E-02	5.62E-01	1.48E+02	1.67E-01	7.18E-01	3.44E-03
2-methyl fluorene	7.43E-04	5.62E-01	1.16E+00	1.31E-03	1.12E-02	1.14E-04

CHEMICAL	Absorp. Factor AF (data input)* (unitless)	Transfer Coefficient FI = (AF*Life) (d)	Final Cattle concentration Cfc = (TID+ID)*FI (ug/kg)	Final Cattle concentration (ppm) conversion = Cfc/1000 (ug/g)
Benzene	1.00E+00	5.48E+02	1.98E+01	1.98E-02
Dioxins	3.70E-01	2.03E+02	4.31E-05	4.31E-08
Mercury	2.50E-02	1.37E+01	9.89E+00	9.89E-03
2-methyl fluorene	1.00E+00	5.48E+02	6.20E+00	6.20E-03

GLC = 24 Hr. average maximum ground level concentrations utilized

WI = water ingestion rate for cattle = 100 kg/d

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

%G = percentage of cattle diet as grazed vegetation (grass and leafy plants) = 67%

%F = percentage of cattle diet as feed = 33%

Flg = grazed vegetation ingestion rate for cattle = 16 kg/d

Flf = feed ingestion rate for cattle = 8 kg/d

%Sf = soil ingestion as percentage of feed ingested by cattle = 1% (0.01)

%Sp = soil ingestion as percentage of grazed vegetation ingested by cattle = 5% (0.05)

RR = respiration rate for cattle = 80 m3/d

* AF = 1 - absorption assumed to be 100% where no data were available for chemical

Life = average life expectancy of cattle = 548 days (1.5 yrs)

BW = average body weight of cattle = 500 kg

Incinerator-field vole model

CHEMICAL	Ground Level conc. GLC (data input) (ug/m3)	Water concentration Cw (data input) (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vegetation conc. Cfv (data input) (ug/kg)	Veg. Ingest. Dose VID = (Cfv*%G*Flg)/BW (ug/kg/d)	Soil concentration Cs (data input) (ug/kg)	Soil Ingest. Dose SID = (Cs*SI)/BW (ug/kg/d)
Benzene	1.30E-01	0.00E+00	0.00E+00	3.66E-03	1.19E-03	1.36E+01	4.41E-02
Dioxins/furans	1.38E-08	0.00E+00	0.00E+00	4.61E-06	1.50E-06	9.20E-05	2.99E-07
Mercury	2.15E-02	0.00E+00	0.00E+00	2.38E+01	7.73E+00	1.48E+02	4.82E-01
2-methyl fluorene	7.14E-04	0.00E+00	0.00E+00	4.27E-01	1.39E-01	1.16E+00	3.77E-03

CHEMICAL	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)	Absorp. Factor AF (data input)* (unitless)	Transfer Coefficient Fi = AF* Life (d)	Final Vole concentration Cfvo = (TID+ID)*Fi (ug/kg)	Final Vole concentration (ppm) conversion = Cfvo/1000 (ug/g)
Benzene	1.69E-01	1.00E+00	9.00E+01	1.93E+01	1.93E-02
Dioxins/furans	1.79E-08	3.70E-01	3.33E+01	6.04E-05	6.04E-08
Mercury	2.80E-02	2.50E-02	2.25E+00	1.85E+01	1.85E-02
2-methyl fluorene	9.28E-04	1.00E+00	9.00E+01	1.29E+01	1.29E-02

GLC = 24 Hr. average maximum ground level concentration utilized

WI = water ingestion rate for field vole (40 g b.w.) = 0.008 kg/d

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

BW = Body weight (adult) 40 g (0.04kg)

%G = percentage of vole diet as grazed vegetation (grass and leafy plants) assumed to be 100%

Flg = grazed vegetation ingestion rate for field vole (40 g b.w.) = 13 g/d (0.013 kg/d)

SI = Soil ingestion rate for field vole = 0.13 g/d (1.3E-4 kg/d) - assumed to be 1% of Flg

RR = respiration rate for field vole (adult male) = 0.052 m3/d

* AF = 1 - absorption assumed to be 100% where no data were available for the chemical.

Life = average life expectancy for field vole = 90 days (3 mo.)

Incinerator-vole-fox model

CHEMICAL	Ground Level conc. GLC (data input) (ug/m3)	Water Concentration Cw (data input) (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vole Concentration Cfvo (data input) (ug/kg)	Vole Ingest. Dose VoID = (Cfv*FI)/BW (ug/kg/d)	Soil Concentration Cs (data input) (ug/kg)	Soil Ingest. Dose SID = (Cs*SI)/BW (ug/kg/d))
Benzene	1.30E-01	0.00E+00	0.00E+00	1.93E+01	1.35E+00	1.36E+01	9.50E-03
Dioxins/furans	1.38E-08	0.00E+00	0.00E+00	6.04E-05	4.23E-06	9.20E-05	6.44E-08
Mercury	2.15E-02	0.00E+00	0.00E+00	1.85E+01	1.30E+00	1.48E+02	1.04E-01
2-methyl fluorene	7.14E-04	0.00E+00	0.00E+00	1.29E+01	9.03E-01	1.16E+00	8.12E-04

CHEMICAL	Total Ingestion Dose TID = WID+VoID+SID (ug/kg/d)	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)	Absorp. Factor AF (data input)* (unitless)	Transfer Coefficient Fi = AF*Life (d)	Final Fox Concentration Cff = (TID+ID)*Fi (ug/kg)	Final Fox Concentration (ppm) conversion = Cff/1000 (ug/g)
Benzene	1.36E+00	5.20E-02	1.00E+00	1.10E+02	1.55E+02	1.55E-01
Dioxins/furans	4.30E-06	5.52E-09	1.00E+00	1.10E+02	4.71E-04	4.71E-07
Mercury	1.40E+00	8.60E-03	1.00E+00	1.10E+02	1.55E+02	1.55E-01
2-methyl fluorene	9.04E-01	2.86E-04	1.00E+00	1.10E+02	9.91E+01	9.91E-02

GLC = 24 Hr. average maximum ground level concentration utilized.

WI = water ingestion rate for red fox (5 kg adult) = 0.42 kg/d

%SW = percent of water ingested from impacted water body - assumed to be 100% in this case.

BW = average body weight for red fox (adult male) = 5.0 kg

FI = vole ingestion rate for red fox (non-breeding 5 kg adult) = 350 g/d (0.35 kg/d)

SI = soil ingestion rate for red fox = 3.5 g/d (0.0035 kg/d) (assumed to be 1% of FI)

RR = respiration rate for red fox (5 kg adult) = 2 m3/d

* AF=1 - absorption assumed to be 100% where no data were available for a chemical.

Life = average life expectancy for red fox = 548 days (1.5 yrs)

Incinerator-vole-hawk model

CHEMICAL	Ground Level conc. GLC (data input) (ug/m3)	Water Concentration Cw (data input) (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vole Concentration Cfvo (data input) (ug/kg)	Vole Ingest. Dose VoID = (Cfv*FI)/BW (ug/kg/d)	Soil Concentration Cs (data input) (ug/kg)	Soil Ingest. Dose SID = (Cs*SI)/BW (ug/kg/d)
Benzene	1.30E-01	0.00E+00	0.00E+00	1.93E+01	1.93E+00	1.36E+01	1.36E-02
Dioxins/furans	1.38E-08	0.00E+00	0.00E+00	6.04E-05	6.04E-06	9.20E-05	9.20E-08
Mercury	2.15E-02	0.00E+00	0.00E+00	1.85E+01	1.85E+00	1.48E+02	1.48E-01
2-methy fluorene	7.14E-04	0.00E+00	0.00E+00	1.29E+01	1.29E+00	1.16E+00	1.16E-03

CHEMICAL	Total Ingestion Dose TID = WID+VoID+SID (ug/kg/d)	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)	Absorp. Factor AF (data input)* (unitless)	Transfer Coefficient Fi = AF*Life (d)	Final Hawk Concentration Cfh = (TID+ID)*Fi (ug/kg)	Final Hawk Concentration (ppm) conversion = Cfh/1000 (ug/g)
Benzene	1.94E+00	1.95E-01	1.00E+00	3.29E+03	7.02E+03	7.02E+00
Dioxins/furans	6.14E-06	2.07E-08	1.00E+00	3.29E+03	2.02E-02	2.02E-05
Mercury	2.00E+00	3.23E-02	1.00E+00	3.29E+03	6.68E+03	6.68E+00
2-methy fluorene	1.29E+00	1.07E-03	1.00E+00	3.29E+03	4.25E+03	4.25E+00

GLC = 24 Hr. average maximum ground level concentration utilized.

WI = water ingestion rate for redtail hawk (1.2 kg adult) = 70 g/d (0.07 kg/d)

%SW = % of water ingested from impacted water body - assumed to be 100% in this case.

BW = average body weight of redtail hawk (adult female) = 1.2 kg

FI = vole ingestion rate for redtail hawk (1.2 kg adult) = 120 g/d (0.12 kg/d)

SI = soil ingestion rate for redtail hawk = 1.2 g/d (0.0012 kg/d) (assumed to be 1% of FI)

RR = respiration rate for redtail hawk (adult female) = 1.8 m3/d

* AF=1 - absorption assumed to be 100% where no data were available for a chemical.

Life = average life expectancy of redtail hawk = 3285 days (9 yrs)

Landfill-vegetation model

CHEMICAL	Ground Level Concentration GLC (data input)* (ug/m3)	Deposition to Vegetation DEP = (GLC*Deprate*sec/d) (ug/m2/d)	Henry's Law Constant H (data input) (Torr.-L/mol-kelvin)	Weathering Constant Rv = Vda/Mf*H/Rte*1000/(0.9+(0.1*Kow))
Benzene	2.62E-01	7.91E+01	4.16E+03	2.10E+03
Bromodichloromethane	1.66E+00	5.01E+02	1.59E+03	1.12E+03
1,2-dibromoethane	6.48E-03	1.96E+00	2.40E+02	1.58E+02
1,1-dichloroethane	4.68E+00	1.42E+03	4.09E+03	4.43E+03
1,2-dichloroethane	5.40E-02	1.63E+01	8.25E+02	1.72E+03
1,1-dichloroethylene	3.57E-01	1.08E+02	1.12E+04	8.40E+03
Ethyl mercaptan	1.64E+00	4.96E+02		
Methyl mercaptan	2.52E+00	7.62E+02		
Methylene chloride	1.10E+01	3.32E+03	2.39E+03	6.70E+03
1,1,2,2-tetrachloroethane	1.66E-01	5.01E+01	2.85E+02	5.86E+01
1,1,2-trichloroethane	6.48E-02	1.96E+01	6.83E+02	3.58E+02
Trichloroethylene	1.27E+00	3.84E+02	6.84E+03	1.03E+03
Vinyl chloride	7.74E-01	2.34E+02	2.09E+04	7.36E+04

CHEMICAL	Deposition concentration vegetation Cdepv = [DEP*IF/(Rv*Y)]*[1-EXP(-RvT)] (ug/kg)	Bioavailability Factor BIO (data input)	Soil half-life t1/2 (data input) (d)	Soil Elimination Ks = (0.693/t1/2)	Integral Function X = [(EXP(-Ks*Tf)-EXP(-Ks*To))/Ks]+Tt
Benzene	7.36E-03	1.00E+00	1.60E+01	4.33E-02	7.28E+03
Bromodichloromethane	8.70E-02	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,2-dibromoethane	2.41E-03	1.00E+00	3.65E+03	1.90E-04	3.35E+03
1,1-dichloroethane	6.24E-02	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,2-dichloroethane	1.85E-03	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,1-dichloroethylene	2.51E-03	1.00E+00	1.80E+02	3.85E-03	7.04E+03
Ethyl mercaptan					
Methyl mercaptan					
Methylene chloride	9.66E-02	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,1,2,2-tetrachloroethane	1.67E-01	1.00E+00	1.00E+01	6.93E-02	7.29E+03
1,1,2-trichloroethane	1.07E-02	1.00E+00	3.65E+02	1.90E-03	6.77E+03
Trichloroethylene	7.28E-02	1.00E+00	1.00E+01	6.93E-02	7.29E+03
Vinyl chloride	6.20E-04	1.00E+00	1.00E+01	6.93E-02	7.29E+03

Landfill-vegetation model

CHEMICAL	Soil conc. Cs = (DEP*X/(Ks*SD*BD*Tt) (ug/kg)	Octanol:water Kow (data input)**	Carbon part. coeff. Koc (data input)**	Uptake Factor UF2 = [(0.03*$Kow^{0.77}$)+0.82]/[(Koc)(Foc)]	Translocation conc. vegetation Ctrans = (Cs*UF2)*** (ug/kg)
Benzene	2.73E+01	1.52E+02	8.90E+01	2.53E-01	0.00E+00
Bromodichloromethane	5.37E+02	1.06E+02	1.52E+02	1.26E-01	0.00E+00
1,2-dibromoethane	7.11E+01	1.14E+02	4.70E+01	4.19E-01	0.00E+00
1,1-dichloroethane	1.52E+03	6.60E+01	3.90E+01	4.04E-01	0.00E+00
1,2-dichloroethane	1.75E+01	3.00E+01	2.40E+01	5.13E-01	0.00E+00
1,1-dichloroethylene	4.06E+02	9.90E+01	1.08E+02	1.71E-01	0.00E+00
Ethyl mercaptan		1.50E+01	9.50E+01	1.12E-01	0.00E+00
Methyl mercaptan		4.50E+00	4.90E+01	1.87E-01	0.00E+00
Methylene chloride	3.56E+03	2.00E+01	1.80E+01	6.23E-01	0.00E+00
1,1,2,2-tetrachloroethane	1.08E+01	3.86E+02	8.70E+01	4.33E-01	0.00E+00
1,1,2-trichloroethane	1.44E+02	1.46E+02	6.30E+01	3.51E-01	0.00E+00
Trichloroethylene	8.30E+01	5.31E+02	2.36E+02	1.94E-01	0.00E+00
Vinyl chloride	5.06E+01	1.40E+01	3.10E+01	3.38E-01	0.00E+00

CHEMICAL	Final vegetation concentration Cfv = (Cdepv*BIO)+Ctrans (ug/kg)	Final vegetation concentration (ppm) conversion = Cfv/1000 (ug/g)
Benzene	7.36E-03	7.36E-06
Bromodichloromethane	8.70E-02	8.70E-05
1,2-dibromoethane	2.41E-03	2.41E-06
1,1-dichloroethane	6.24E-02	6.24E-05
1,2-dichloroethane	1.85E-03	1.85E-06
1,1-dichloroethylene	2.51E-03	2.51E-06
Ethyl mercaptan	0.00E+00	0.00E+00
Methyl mercaptan	0.00E+00	0.00E+00
Methylene chloride	9.66E-02	9.66E-05
1,1,2,2-tetrachloroethane	1.67E-01	1.67E-04
1,1,2-trichloroethane	1.07E-02	1.07E-05
Trichloroethylene	7.28E-02	7.28E-05
Vinyl chloride	6.20E-04	6.20E-07

Landfill-vegetation model

* Max. ground level concentration (24 Hr. average) 17-20 utilized (year operation period)
Deprate = deposition rate of 0.0035 m/s was utilized for volatiles.
sec/d = number of seconds per day = 86,400
IF = interception fraction for vegetation (grass and leafy plants) = 0.39
Rv = weathering constant for volatile contaminants (1/d)
Vda = 302 m/d
Mf = vegetation yield = 2.0 kg/m2
H = Henry's Law constant (Torr.-L/mol-kelvin) = (Pa M3/mol x 7.5)
R = Gas constant = 62.4 Torr.-L/mol-kelvin
Te = temperature = 298°K
Kow = octanol:water constant
Y = seasonal vegetation yield (grass and leafy plants) = 2 kg/m2
EXP = exp base e = 2.72
T = growth period for vegetation (grass and leafy plants) = 90 days
To = beginning of evaluation period = 0 days
Tf = end of evaluation period = 7300 days (20 year life expectancy of facility.
Tt = total exposure period (tf-To) = 7300 days (20 year life expectancy of facility.
SD = soil mixing depth = 0.05 m
BD = soil bulk density = 1,333 kg/m3

** Kow and Koc values for fluorene used for 2-methyl fluorene
For UF2, 0.03, 0.77 and 0.82 are empirical constants
*** For volatiles, the uptake from soil through roots is assumed to be zero.
Foc = fraction organic carbon in soil = 0.1%

Landfill-cattle model

CHEMICAL	Ground Level Concentration GLC (input)* (ug/m3)	Water concentration Cw (data input)** (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vegetation conc. Cfv (data input) (ug/kg)	Final Feed conc. Cff (data input) (ug/kg)	Vegetation Ingestion Dose VID = (Cfg*%G*Flg)/BW (ug/kg/d)
Benzene	2.62E-01	2.60E-02	5.20E-03	7.36E-03	7.36E-03	1.58E-04
Bromodichloromethane	1.66E+00	3.10E-02	6.20E-03	8.70E-02	8.70E-02	1.86E-03
1,2-dibromoethane	6.48E-03		0.00E+00	2.41E-03	2.41E-03	5.17E-05
1,1-dichloroethane	4.68E+00	4.80E+00	9.60E-01	6.24E-02	6.24E-02	1.34E-03
1,2-dichloroethane	5.40E-02	2.80E-02	5.60E-03	1.85E-03	1.85E-03	3.97E-05
1,1-dichloroethylene	3.57E-01	4.20E-02	8.40E-03	2.51E-03	2.51E-03	5.37E-05
ethyl mercaptan	1.64E+00		0.00E+00	0.00E+00	0.00E+00	0.00E+00
methyl mercaptan	2.52E+00		0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.10E+01	5.00E-02	1.00E-02	9.66E-02	9.66E-02	2.07E-03
1,1,2,2-tetrachloroethane	1.66E-01	1.80E-02	3.60E-03	1.67E-01	1.67E-01	3.57E-03
1,1,2-trichloroethane	6.48E-02	3.00E-02	6.00E-03	1.07E-02	1.07E-02	2.29E-04
Trichloroethylene	1.27E+00	4.30E-04	8.60E-05	7.28E-02	7.28E-02	1.56E-03
vinyl chloride	7.74E-01	1.71E-01	3.42E-02	6.20E-04	6.20E-04	1.33E-05

CHEMICAL	Feed Ingestion Dose FID = [(1-%G)*Flf*%F*Cff]/BW (ug/kg/d)	Soil Ingestion Rate SI = [(1-%G)*%Sf*Flf]+%G*%Sp*Flg (ug/kg)	Soil Concentration Cs (data input) (ug/kg)	Soil Ingestion Dose SID = (Cs*SI)/BW (ug/kg/d)	Total Ingestion Dose TID =WID+VID+FID+SID (ug/kg/d)
Benzene	1.28E-05	5.62E-01	2.73E+01	3.07E-02	3.61E-02
Bromodichloromethane	1.52E-04	5.62E-01	5.37E+02	6.04E-01	6.12E-01
1,2-dibromoethane	4.20E-06	5.62E-01	7.11E+01	7.99E-02	8.00E-02
1,1-dichloroethane	1.09E-04	5.62E-01	1.52E+03	1.71E+00	2.67E+00
1,2-dichloroethane	3.23E-06	5.62E-01	1.75E+01	1.97E-02	2.53E-02
1,1-dichloroethylene	4.37E-06	5.62E-01	4.06E+02	4.56E-01	4.65E-01
ethyl mercaptan	0.00E+00	5.62E-01	0.00E+00	0.00E+00	0.00E+00
methyl mercaptan	0.00E+00	5.62E-01	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.68E-04	5.62E-01	3.56E+03	4.00E+00	4.02E+00
1,1,2,2-tetrachloroethane	2.90E-04	5.62E-01	1.08E+01	1.22E-02	1.96E-02
1,1,2-trichloroethane	1.86E-05	5.62E-01	1.44E+02	1.62E-01	1.68E-01
Trichloroethylene	1.27E-04	5.62E-01	8.30E+01	9.33E-02	9.51E-02
vinyl chloride	1.08E-06	5.62E-01	5.06E+01	5.69E-02	9.11E-02

Landfill-cattle model

CHEMICAL	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)	Absorption Factor AF (data input)*** (unitless)	Transfer Coefficient FI = AF*Life (d)	Final Cattle concentration Cfc = (TID+ID)*FI (ug/kg)	Final Cattle Concentration (ppm) conversion = Cfc/1000 (ug/g)
Benzene	4.19E-02	1.00E+00	5.48E+02	4.27E+01	4.27E-02
Bromodichloromethane	2.65E-01	1.00E+00	5.48E+02	4.81E+02	4.81E-01
1,2-dibromoethane	1.04E-03	1.00E+00	5.48E+02	4.44E+01	4.44E-02
1,1-dichloroethane	7.49E-01	1.00E+00	5.48E+02	1.87E+03	1.87E+00
1,2-dichloroethane	8.64E-03	1.00E+00	5.48E+02	1.86E+01	1.86E-02
1,1-dichloroethylene	5.71E-02	1.00E+00	5.48E+02	2.86E+02	2.86E-01
ethyl mercaptan	2.62E-01	1.00E+00	5.48E+02	1.44E+02	1.44E-01
methyl mercaptan	4.03E-01	1.00E+00	5.48E+02	2.21E+02	2.21E-01
Methylene chloride	1.76E+00	1.00E+00	5.48E+02	3.16E+03	3.16E+00
1,1,2,2-tetrachloroethane	2.65E-02	1.00E+00	5.48E+02	2.53E+01	2.53E-02
1,1,2-trichloroethane	1.04E-02	1.00E+00	5.48E+02	9.77E+01	9.77E-02
Trichloroethylene	2.03E-01	1.00E+00	5.48E+02	1.63E+02	1.63E-01
vinyl chloride	1.24E-01	1.00E+00	5.48E+02	1.18E+02	1.18E-01

* Maximum ground level concentration (24 Hr. average) utilized (17-20 year operation period).

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for cattle = 100 kg/d

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

BW = average body weight of cattle = 500 kg

%G = percentage of cattle diet as grazed vegetation (grass and leafy plants) = 67% (0.67)

%F = percentage of cattle diet as feed = 33% (0.33)

FIg = grazed vegetation ingestion rate for cattle = 16 kg/d

FIf = feed ingestion rate for cattle = 8 kg/d

%Sf = soil ingestion as percentage of feed ingested by cattle = 1% (0.01)

% Sp = soil ingestion as percentage of grazed vegetation ingested by cattle = 5% (0.05)

RR = respiration rate for cattle = 80 m3/d

*** AF=1 - absorption assumed to be 100% where no data were available for a chemical.

Life = average life expectancy of cattle = 548 days (1.5 yrs)

Landfill-field vole model

CHEMICAL	Ground Level conc. GLC (data input)* (ug/m3)	Water conc. Cw (data input)** (ug/g)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vegetation conc. Cfv (data input) (ug/kg)	Veg. Ingest. Dose VID = (Cfg*%G*Flg)/BW (ug/kg/d)	Soil Conc. Cs (data input) (ug/kg)	Soil Ingestion Dose SID = (Cs*SI)/BW (ug/kg/d)
Benzene	2.62E-01	2.60E-02	5.20E-03	7.36E-03	2.39E-03	2.73E+01	8.88E-02
Bromodichloromethane	1.66E+00	3.10E-02	6.20E-03	8.70E-02	2.83E-02	5.37E+02	1.74E+00
1,2-dibromoethane	6.48E-03	0.00E+00	0.00E+00	2.41E-03	7.84E-04	7.11E+01	2.31E-01
1,1-dichloroethane	4.68E+00	4.80E+00	9.60E-01	6.24E-02	2.03E-02	1.52E+03	4.93E+00
1,2-dichloroethane	5.40E-02	2.80E-02	5.60E-03	1.85E-03	6.02E-04	1.75E+01	5.69E-02
1,1-dichloroethylene	3.57E-01	4.20E-02	8.40E-03	2.51E-03	8.14E-04	4.06E+02	1.32E+00
Ethyl mercaptan	1.64E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methyl mercaptan	2.52E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.10E+01	5.00E-02	1.00E-02	9.66E-02	3.14E-02	3.56E+03	1.16E+01
1,1,2,2-tetrachloroethane	1.66E-01	1.80E-02	3.60E-03	1.67E-01	5.42E-02	1.08E+01	3.52E-02
1,1,2-trichloroethane	6.48E-02	3.00E-02	6.00E-03	1.07E-02	3.47E-03	1.44E+02	4.67E-01
Trichloroethylene	1.27E+00	4.30E-04	8.60E-05	7.28E-02	2.37E-02	8.30E+01	2.70E-01
Vinyl chloride	7.74E-01	1.71E-01	3.42E-02	6.20E-04	2.02E-04	5.06E+01	1.64E-01

* Maximum ground level concentration (24 Hr. average) utilized (17-20 year operation period).

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for field vole (40 g adult) = 0.008 kg/d

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

BW = average body weight (adult) = 40 g (0.04 kg)

%G = percentage of vole diet as grazed vegetation (grass and leafy plants) - assumed to be 100%.

Flg = grazed vegetation ingestion rate for field vole (40 g adult) = 13 g/d (0.013 kg/d)

RR = respiration rate for field vole (adult male) = 0.052 m3/d

SI = soil ingestion rate for field vole = 0.13 g/d (1.3E-4 kg/d) - assumed to be 1% of Flg

***AF=1 - absorption assumed to be 100% where no data was available for chemical.

Life = average life expectancy for field vole = 90 days (3 mo.)

Landfill-field vole model

CHEMICAL	Total Ingestion Dose TID = WID+VID+SID (ug/kg/d)	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)	Absorp. Factor AF (data input)*** (unitless)	Transfer Coefficient Fi = AF*Life (d)	Final Vole conc. Cfvo =(TID+ID)*Fi (ug/kg)	Final Vole conc. (ppm) conversion = Cfvo/1000 (ug/g)
Benzene	9.64E-02	3.40E-01	1.00E+00	9.00E+01	3.93E+01	3.93E-02
Bromodichloromethane	1.78E+00	2.15E+00	1.00E+00	9.00E+01	3.54E+02	3.54E-01
1,2-dibromoethane	2.32E-01	8.42E-03	1.00E+00	9.00E+01	2.16E+01	2.16E-02
1,1-dichloroethane	5.91E+00	6.08E+00	1.00E+00	9.00E+01	1.08E+03	1.08E+00
1,2-dichloroethane	6.31E-02	7.02E-02	1.00E+00	9.00E+01	1.20E+01	1.20E-02
1,1-dichloroethylene	1.33E+00	4.64E-01	1.00E+00	9.00E+01	1.61E+02	1.61E-01
Ethyl mercaptan	0.00E+00	2.13E+00	1.00E+00	9.00E+01	1.92E+02	1.92E-01
Methyl mercaptan	0.00E+00	3.28E+00	1.00E+00	9.00E+01	2.95E+02	2.95E-01
Methylene chloride	1.16E+01	1.43E+01	1.00E+00	9.00E+01	2.33E+03	2.33E+00
1,1,2,2-tetrachloroethane	9.29E-02	2.15E-01	1.00E+00	9.00E+01	2.77E+01	2.77E-02
1,1,2-trichloroethane	4.76E-01	8.42E-02	1.00E+00	9.00E+01	5.05E+01	5.05E-02
Trichloroethylene	2.93E-01	1.65E+00	1.00E+00	9.00E+01	1.75E+02	1.75E-01
Vinyl chloride	1.99E-01	1.01E+00	1.00E+00	9.00E+01	1.08E+02	1.08E-01

Landfill-vole-fox model

CHEMICAL	Ground Level conc. GLC (data input)* (ug/m3)	Water conc. Cw (data input)** (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vole Conc. Cfvo (data input) (ug/kg)	Vole Ingest. Dose VoID = (Cfv*FI)/BW (ug/kg/d)	Soil Conc. Cs (data input) (ug/kg)	Soil Ingestion Dose SID = (Cs*SI)/BW (ug/kg/d)
Benzene	2.62E-01	2.60E-02	2.18E-03	3.93E+01	2.75E+00	2.73E+01	1.91E-02
Bromodichloroethane	1.66E+00	3.10E-02	2.60E-03	3.54E+02	2.48E+01	5.37E+02	3.76E-01
1,2-dibromoethane	6.48E-03	0.00E+00	0.00E+00	2.16E+01	1.51E+00	7.11E+01	4.97E-02
1,1-dichloroethane	4.68E+00	4.80E+00	4.03E-01	1.08E+03	7.56E+01	1.52E+03	1.06E+00
1,2-dichloroethane	5.40E-02	2.80E-02	2.35E-03	1.20E+01	8.40E-01	1.75E+01	1.23E-02
1,1-dichloroethylene	3.57E-01	4.20E-02	3.53E-03	1.61E+02	1.13E+01	4.06E+02	2.84E-01
Ethyl mercaptan	1.64E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methyl mercaptan	2.52E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.10E+01	5.00E-02	4.20E-03	2.33E+03	1.63E+02	3.56E+03	2.49E+00
1,1,2,2-tetrachloroethane	1.66E-01	1.80E-02	1.51E-03	2.77E+01	1.94E+00	1.08E+01	7.57E-03
1,1,2-trichloroethane	6.48E-02	3.00E-02	2.52E-03	5.05E+01	3.53E+00	1.44E+02	1.01E-01
Trichloroethylene	1.27E+00	4.30E-04	3.61E-05	1.75E+02	1.23E+01	8.30E+01	5.81E-02
Vinyl chloride	7.74E-01	1.71E-01	1.44E-02	1.08E+02	7.59E+00	5.06E+01	3.54E-02

* Maximum ground level concentration (24 Hr. average) utilized (17-20 year operation period).

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for red fox (5 kg adult) = 420 g/d (0.42 kg/d)

%SW = percent of water ingested from impacted water body -assumed to be 100% in this case.

BW = average body weight for red fox (adult male) = 5.0 kg

FI = vole ingestion rate for red fox (non-breeding 5 kg adult) = 350 g/d (0.35 kg/d)

SI = soil ingestion rate for red fox = 3.53 g/d (0.0035 kg/d) - assumed to be 1% of FI.

RR = respiration rate for red fox (5 kg adult) = 2 m3/d

*** AF=1 - absorption assumed to be 100% where no data was available for a chemical.

Life = average life expectancy for red fox = 548 days (1.5 yrs).

Landfill-vole-fox model

CHEMICAL	Total Ingestion Dose TID = WID+VoID+SID (ug/kg/d)	Total Inhalation Dose ID = (GLC*RR)/BW (ug/d)	Absorp. Factor AF (data input)*** (unitless)	Transfer Coefficient Fi = AF*Life (d)	Final Fox Conc. Cff = (TID+ID)*Fi (ug/kg)	Final Fox conc. (ppm) conversion = Cff/1000 (ug/g)
Benzene	2.77E+00	1.05E-01	1.00E+00	5.48E+02	1.58E+03	1.58E+00
Bromodichloroethane	2.51E+01	6.62E-01	1.00E+00	5.48E+02	1.41E+04	1.41E+01
1,2-dibromoethane	1.56E+00	2.59E-03	1.00E+00	5.48E+02	8.58E+02	8.58E-01
1,1-dichloroethane	7.70E+01	1.87E+00	1.00E+00	5.48E+02	4.32E+04	4.32E+01
1,2-dichloroethane	8.54E-01	2.16E-02	1.00E+00	5.48E+02	4.80E+02	4.80E-01
1,1-dichloroethylene	1.16E+01	1.43E-01	1.00E+00	5.48E+02	6.42E+03	6.42E+00
Ethyl mercaptan	0.00E+00	6.56E-01	1.00E+00	5.48E+02	3.59E+02	3.59E-01
Methyl mercaptan	0.00E+00	1.01E+00	1.00E+00	5.48E+02	5.52E+02	5.52E-01
Methylene chloride	1.66E+02	4.39E+00	1.00E+00	5.48E+02	9.31E+04	9.31E+01
1,1,2,2-tetrachloroethane	1.95E+00	6.62E-02	1.00E+00	5.48E+02	1.11E+03	1.11E+00
1,1,2-trichloroethane	3.64E+00	2.59E-02	1.00E+00	5.48E+02	2.01E+03	2.01E+00
Trichloroethylene	1.23E+01	5.08E-01	1.00E+00	5.48E+02	7.02E+03	7.02E+00
Vinyl chloride	7.64E+00	3.10E-01	1.00E+00	5.48E+02	4.36E+03	4.36E+00

Landfill-vole-hawk model

CHEMICAL	Ground Level conc. GLC (data input)* (ug/m3)	Water conc. Cw (data input)** (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)/BW (ug/kg/d)	Final Vole conc. Cfvo (data input) (ug/d)	Vole Ingest. Dose VoID = (Cfv*FI)/BW (ug/kg/d)	Soil conc. Cs (data input) (ug/kg)	Soil Ingestion Dose SID = (Cs*SI)/BW (ug/d)
Benzene	2.62E-01	2.60E-02	1.52E-03	3.93E+01	3.93E+00	2.73E+01	2.73E-02
Bromodichloroethane	1.66E+00	3.10E-02	1.81E-03	3.54E+02	3.54E+01	5.37E+02	5.37E-01
1,2-dibromoethane	6.48E-03	0.00E+00	0.00E+00	2.16E+01	2.16E+00	7.11E+01	7.11E-02
1,1-dichloroethane	4.68E+00	4.80E+00	2.80E-01	1.08E+03	1.08E+02	1.52E+03	1.52E+00
1,2-dichloroethane	5.40E-02	2.80E-02	1.63E-03	1.20E+01	1.20E+00	1.75E+01	1.75E-02
1,1-dichloroethylene	3.57E-01	4.20E-02	2.45E-03	1.61E+02	1.61E+01	4.06E+02	4.06E-01
Ethyl mercaptan	1.64E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methyl mercaptan	2.52E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.10E+01	5.00E-02	2.92E-03	2.33E+03	2.33E+02	3.56E+03	3.56E+00
1,1,2,2-tetrachloroethane	1.66E-01	1.80E-02	1.05E-03	2.77E+01	2.77E+00	1.08E+01	1.08E-02
1,1,2-trichloroethane	6.48E-02	3.00E-02	1.75E-03	5.05E+01	5.05E+00	1.44E+02	1.44E-01
Trichloroethylene	1.27E+00	4.30E-04	2.51E-05	1.75E+02	1.75E+01	8.30E+01	8.30E-02
Vinyl chloride	7.74E-01	1.71E-01	9.98E-03	1.08E+02	1.08E+01	5.06E+01	5.06E-02

* Maximum ground level concentration (24 Hr. average) utilized (17-20 year operation period).

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for redtail hawk (1.2 kg adult) = 70 g/d (0.07 kg/d)

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

BW = average body weight of redtail hawk (adult female) = 1.2 kg

FI = vole ingestion rate for redtail hawk (1.2 kg adult) = 120 g/d (0.12 kg/d)

SI = soil ingestion rate for redtail hawk = 0.12 g/d (0.0012 kg/d) - assumed to be 1% of FI.

RR = respiration rate for redtail hawk (adult female) = 1.8 m3/d

*** AF=1 - absorption assumed to be 100% where no data were available for a chemical.

Life = average life expectancy of redtail hawk = 3285 days (9 yrs).

Landfill-vole-hawk model

CHEMICAL	Total Ingestion Dose TID = WID+VoID+SID (ug/kg/d)	Total Inhalation Dose ID = (GLC*RR)/BW (ug/kg/d)	Absorp. Factor AF (data input)*** (unitless)	Transfer Coefficient Fi = AF*Life (d)	Final Hawk conc. Cfh = (TID+ID)*FI (ug/kg)	Final Hawk conc. (ppm) conversion - Cfh/1000 (ug/g)
Benzene	3.96E+00	3.92E-01	1.00E+00	2.74E+03	1.19E+04	1.19E+01
Bromodichloroethane	3.59E+01	2.48E+00	1.00E+00	2.74E+03	1.05E+05	1.05E+02
1,2-dibromoethane	2.23E+00	9.72E-03	1.00E+00	2.74E+03	6.14E+03	6.14E+00
1,1-dichloroethane	1.10E+02	7.02E+00	1.00E+00	2.74E+03	3.20E+05	3.20E+02
1,2-dichloroethane	1.22E+00	8.10E-02	1.00E+00	2.74E+03	3.56E+03	3.56E+00
1,1-dichloroethylene	1.65E+01	5.36E-01	1.00E+00	2.74E+03	4.67E+04	4.67E+01
Ethyl mercaptan	0.00E+00	2.46E+00	1.00E+00	2.74E+03	6.73E+03	6.73E+00
Methyl mercaptan	0.00E+00	3.78E+00	1.00E+00	2.74E+03	1.03E+04	1.03E+01
Methylene chloride	2.36E+02	1.65E+01	1.00E+00	2.74E+03	6.92E+05	6.92E+02
1,1,2,2-tetrachloroethane	2.79E+00	2.48E-01	1.00E+00	2.74E+03	8.31E+03	8.31E+00
1,1,2-trichloroethane	5.19E+00	9.72E-02	1.00E+00	2.74E+03	1.45E+04	1.45E+01
Trichloroethylene	1.76E+01	1.91E+00	1.00E+00	2.74E+03	5.33E+04	5.33E+01
Vinyl chloride	1.09E+01	1.16E+00	1.00E+00	2.74E+03	3.30E+04	3.30E+01

Landfill-vegetation model (2)

CHEMICAL	Ground Level Concentration GLC (data input)* (ug/m3)	Deposition to Vegetation DEP = (GLC*Deprate*sec/d) (ug/m2/d)	Henry's Law Constant H (data input) (Torr.-L/mol-kelvin)	Weathering Constant Rv = Vda/Mf*H/Rte*1000/(0.9+(0.1*Kow))
Benzene	2.62E-01	7.91E+01	4.16E+03	2.10E+03
Bromodichloromethane	1.78E-01	5.38E+01	1.59E+03	1.12E+03
1,2-dibromoethane	6.48E-03	1.96E+00	2.40E+02	1.58E+02
1,1-dichloroethane	5.04E-01	1.52E+02	4.09E+03	4.43E+03
1,2-dichloroethane	5.82E-03	1.76E+00	8.25E+02	1.72E+03
1,1-dichloroethylene	3.88E-02	1.17E+01	1.12E+04	8.40E+03
Ethyl mercaptan	1.76E-01	5.32E+01		
Methyl mercaptan	2.72E-01	8.23E+01		
Methylene chloride	1.18E+00	3.57E+02	2.39E+03	6.70E+03
1,1,2,2-tetrachloroethane	1.80E-02	5.44E+00	2.85E+02	5.86E+01
1,1,2-trichloroethane	6.98E-03	2.11E+00	6.83E+02	3.58E+02
Trichloroethylene	1.38E-01	4.17E+01	6.84E+03	1.03E+03
Vinyl chloride	8.34E-02	2.52E+01	2.09E+04	7.36E+04

CHEMICAL	Bioavailability Factor BIO (data input)	Soil half-life t1/2 (data input) (d)	Soil Elimination Ks = (0.693/t1/2)	Integral Function X = [{EXP(-Ks*Tf)-EXP(-Ks*To)}/Ks]+Tt
Benzene	1.00E+00	1.60E+01	4.33E-02	7.28E+03
Bromodichloromethane	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,2-dibromoethane	1.00E+00	3.65E+03	1.90E-04	3.35E+03
1,1-dichloroethane	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,2-dichloroethane	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,1-dichloroethylene	1.00E+00	1.80E+02	3.85E-03	7.04E+03
Ethyl mercaptan				
Methyl mercaptan				
Methylene chloride	1.00E+00	5.00E+01	1.39E-02	7.23E+03
1,1,2,2-tetrachloroethane	1.00E+00	1.00E+01	6.93E-02	7.29E+03
1,1,2-trichloroethane	1.00E+00	3.65E+02	1.90E-03	6.77E+03
Trichloroethylene	1.00E+00	1.00E+01	6.93E-02	7.29E+03
Vinyl chloride	1.00E+00	1.00E+01	6.93E-02	7.29E+03

Landfill-vegetation model (2)

CHEMICAL	Octanol:water Kow (data input)*	Carbon part. coeff. Koc (data input)*	Uptake Factor UF2 = [(0.03*Kow$^{0.77}$)+0.82]/[(Koc)(Foc)]	Translocation conc. vegetation Ctrans = (Cs*UF2)*** (ug/kg)
Benzene	1.52E+02	8.90E+01	2.53E-01	0.00E+00
Bromodichloromethane	1.06E+02	1.52E+02	1.26E-01	0.00E+00
1,2-dibromoethane	1.14E+02	4.70E+01	4.19E-01	0.00E+00
1,1-dichloroethane	6.60E+01	3.90E+01	4.04E-01	0.00E+00
1,2-dichloroethane	3.00E+01	2.40E+01	5.13E-01	0.00E+00
1,1-dichloroethylene	9.90E+01	1.08E+02	1.71E-01	0.00E+00
Ethyl mercaptan	1.50E+01	9.50E+01	1.12E-01	0.00E+00
Methyl mercaptan	4.50E+00	4.90E+01	1.87E-01	0.00E+00
Methylene chloride	2.00E+01	1.80E+01	6.23E-01	0.00E+00
1,1,2,2-tetrachloroethane	3.86E+02	8.70E+01	4.33E-01	0.00E+00
1,1,2-trichloroethane	1.46E+02	6.30E+01	3.51E-01	0.00E+00
Trichloroethylene	5.31E+02	2.36E+02	1.94E-01	0.00E+00
Vinyl chloride	1.40E+01	3.10E+01	3.38E-01	0.00E+00

CHEMICAL	Final vegetation concentration (ppm) conversion = Cfv/1000 (ug/g)
Benzene	7.36E-06
Bromodichloromethane	9.35E-06
1,2-dibromoethane	2.41E-06
1,1-dichloroethane	6.71E-06
1,2-dichloroethane	2.00E-07
1,1-dichloroethylene	2.72E-07
Ethyl mercaptan	0.00E+00
Methyl mercaptan	0.00E+00
Methylene chloride	1.04E-05
1,1,2,2-tetrachloroethane	1.81E-05
1,1,2-trichloroethane	1.15E-06
Trichloroethylene	7.91E-06
Vinyl chloride	6.68E-08

Landfill-vegetation model (2)

* Maximum ground level concentration (annual average) utilized (17-20 year period of operation)

Deprate = deposition rate of 0.0035 m/s was utilized for volatiles.

sec/d = number of seconds per day = 86,400

IF = interception fraction for vegetation (grass and leafy plants) = 0.39

Rv = weathering constant for volatile contaminants (1/d)

Vda = 302 m/d

Mf = vegetation yield = 2.0 kg/m2

H = Henry's Law constant (Torr.-L/mol-kelvin) = (Pa M3/mol x 7.5)

R = Gas constant = 62.4
Torr.-L/mol-kelvin
Te = temperature = 298°K

Kow = octanol:water constant

Y = seasonal vegetation yield (grass and leafy plants) = 2 kg/m2

EXP = exp base e = 2.72

T = growth period for vegetation (grass and leafy plants) = 90 days

To = beginning of evaluation period = 0 days

Tf = end of evaluation period = 7300 days (20 year life expectancy of facility.

Tt = total exposure period (tf-To) = 7300 days (20 year life expectancy of facility.

SD = soil mixing depth = 0.05 m

BD = soil bulk density = 1,333 kg/m3

Landfill-cattle model (2)

CHEMICAL	Ground Level Concentration GLC (input)* (ug/m3)	Inhalation Dose ID = (GLC*RR) (ug/d)	Water concentration Cw (data input)** (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW)	Final Vegetation conc. Cfv (data input) (ug/kg)	Final Feed conc. Cff (data input) (ug/kg)
Benzene	2.62E-01	2.09E+01	2.60E-02	2.60E+00	7.36E-03	7.36E-03
Bromodichloromethane	1.78E-01	1.42E+01	3.10E-02	3.10E+00	9.35E-03	9.35E-03
1,2-dibromomethane	6.48E-03	5.18E-01		0.00E+00	2.41E-03	2.41E-03
1,1-dichloroethane	5.04E-01	4.03E+01	4.80E+00	4.80E+02	6.71E-03	6.71E-03
1,2-dichloroethane	5.82E-03	4.66E-01	2.80E-02	2.80E+00	2.00E-04	2.00E-04
1,1-dichloroethylene	3.88E-02	3.10E+00	4.20E-02	4.20E+00	2.72E-04	2.72E-04
ethyl mercaptan	1.76E-01	1.41E+01		0.00E+00	0.00E+00	0.00E+00
methyl mercaptan	2.72E-01	2.18E+01		0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.18E+00	9.44E+01	5.00E-02	5.00E+00	1.04E-02	1.04E-02
1,1,2,2-tetrachloroethane	1.80E-02	1.44E+00	1.80E-02	1.80E+00	1.81E-02	1.81E-02
1,1,2-trichloroethane	6.98E-03	5.58E-01	3.00E-02	3.00E+00	1.15E-03	1.15E-03
Trichloroethylene	1.38E-01	1.10E+01	4.30E-04	4.30E-02	7.91E-03	7.91E-03
vinyl chloride	8.34E-02	6.67E+00	1.71E-01	1.71E+01	6.68E-05	6.68E-05

* Maximum ground level concentration (annual average) utilized (17-20 year period of operation).

RR = respiration rate for cattle = 80 m3/d

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for cattle = 100 kg/d

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

%G = percentage of cattle diet as grazed vegetation (grass and leafy plants) = 67% (0.67)

%F = percentage of cattle diet as feed = 33% (0.33)

Flg = grazed vegetation ingestion rate for cattle = 16 kg/d

Flf = feed ingestion rate for cattle = 8 kg/d

%Sf = soil ingestion as percentage of feed ingested by cattle = 1% (0.01)

% Sp = soil ingestion as percentage of grazed vegetation ingested by cattle = 5% (0.05)

*** AF=1 - absorption assumed to be 100% where no data were available for a chemical.

Life = average life expectancy of cattle = 548 days (1.5 yrs)

BW = average body weight of cattle = 500 kg

Landfill-cattle model (2)

CHEMICAL	Vegetation Ingestion Dose VID = (Cfg*%G*Flg) (ug/d)	Feed Ingestion Dose FID = [(1-%G)*Flf*%F*Cff] (ug/d)	Soil Ingestion Rate SI = [(1-%G)*%Sf*Flf]+%G*%Sp*Flg (ug/kg)	Soil Concentration Cs (data input) (ug/kg)	Soil Ingestion Dose SID = (Cs*SI) (ug/d)
Benzene	7.89E-02	6.41E-03	5.62E-01	2.73E+01	1.54E+01
Bromodichloromethane	1.00E-01	8.14E-03	5.62E-01	5.77E+01	3.24E+01
1,2-dibromomethane	2.59E-02	2.10E-03	5.62E-01	7.11E+01	4.00E+01
1,1-dichloroethane	7.20E-02	5.85E-03	5.62E-01	1.63E+02	9.19E+01
1,2-dichloroethane	2.14E-03	1.74E-04	5.62E-01	1.89E+00	1.06E+00
1,1-dichloroethylene	2.92E-03	2.37E-04	5.62E-01	4.41E+01	2.48E+01
ethyl mercaptan	0.00E+00	0.00E+00	5.62E-01	0.00E+00	0.00E+00
methyl mercaptan	0.00E+00	0.00E+00	5.62E-01	0.00E+00	0.00E+00
Methylene chloride	1.11E-01	9.05E-03	5.62E-01	3.82E+02	2.15E+02
1,1,2,2-tetrachloroethane	1.94E-01	1.58E-02	5.62E-01	1.18E+00	6.61E-01
1,1,2-trichloroethane	1.23E-02	1.00E-03	5.62E-01	1.55E+01	8.70E+00
Trichloroethylene	8.48E-02	6.89E-03	5.62E-01	9.02E+00	5.07E+00
vinyl chloride	7.16E-04	5.82E-05	5.62E-01	5.45E+00	3.06E+00

CHEMICAL	Total Exposure Dose TED = ID+WID+VID+FID+SID (ug/d)	Absorption Factor AF (data input)***	Transfer Coefficient FI = (AF*Life/BW) (d/kg)	Final Cattle concentration Cfc = TED*FI (ug/kg)	Final Cattle Concentration (ppm) conversion = Cfc/1000 (ug/g)
Benzene	3.90E+01	1.00E+00	1.10E+00	4.27E+01	4.27E-02
Bromodichloromethane	4.99E+01	1.00E+00	1.10E+00	5.47E+01	5.47E-02
1,2-dibromomethane	4.05E+01	1.00E+00	1.10E+00	4.44E+01	4.44E-02
1,1-dichloroethane	6.12E+02	1.00E+00	1.10E+00	6.71E+02	6.71E-01
1,2-dichloroethane	4.33E+00	1.00E+00	1.10E+00	4.74E+00	4.74E-03
1,1-dichloroethylene	3.21E+01	1.00E+00	1.10E+00	3.52E+01	3.52E-02
ethyl mercaptan	1.41E+01	1.00E+00	1.10E+00	1.54E+01	1.54E-02
methyl mercaptan	2.18E+01	1.00E+00	1.10E+00	2.38E+01	2.38E-02
Methylene chloride	3.15E+02	1.00E+00	1.10E+00	3.45E+02	3.45E-01
1,1,2,2-tetrachloroethane	4.11E+00	1.00E+00	1.10E+00	4.51E+00	4.51E-03
1,1,2-trichloroethane	1.23E+01	1.00E+00	1.10E+00	1.35E+01	1.35E-02
Trichloroethylene	1.62E+01	1.00E+00	1.10E+00	1.78E+01	1.78E-02
vinyl chloride	2.68E+01	1.00E+00	1.10E+00	2.94E+01	2.94E-02

Landfill-field vole model (2)

CHEMICAL	Ground Level concentration GLC (data input)* (ug/m3)	Inhalation Dose ID = (GLC*RR) (ug/d)	Water Concentration Cw (data input)** (ug/g)	Water Ingest. Dose WID = (Cw*WI*%SW) (ug/d)	Final Vegetation conc. Cfv (data input) (ug/kg)	Veg. Ingest. Dose VID = (Cfg*%G*Flg (ug/d)	Soil Concentration Cs (data input) (ug/kg)
Benzene	2.62E-01	1.36E-02	2.60E-02	2.08E-04	7.36E-03	9.57E-05	2.73E+01
Bromodichloromethane	1.78E-01	9.26E-03	3.10E-02	2.48E-04	9.35E-03	1.22E-04	5.77E+01
1,2-dibromoethane	6.48E-03	3.37E-04	0.00E+00	0.00E+00	2.41E-03	3.14E-05	7.11E+01
1,1-dichloroethane	5.04E-01	2.62E-02	4.80E+00	3.84E-02	6.71E-03	8.73E-05	1.63E+02
1,2-dichloroethane	5.82E-03	3.03E-04	2.80E-02	2.24E-04	2.00E-04	2.60E-06	1.89E+00
1,1-dichloroethylene	3.88E-02	2.02E-03	4.20E-02	3.36E-04	2.72E-04	3.54E-06	4.41E+01
Ethyl mercaptan	1.76E-01	9.15E-03	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methyl mercaptan	2.72E-01	1.41E-02	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.18E+00	6.14E-02	5.00E-02	4.00E-04	1.04E-02	1.35E-04	3.82E+02
1,1,2,2-tetrachloroethane	1.80E-02	9.36E-04	1.80E-02	1.44E-04	1.81E-02	2.36E-04	1.18E+00
1,12-trichloroethylene	6.98E-03	3.63E-04	3.00E-02	2.40E-04	1.15E-03	1.50E-05	1.55E+01
Trichloroethylene	1.38E-01	7.18E-03	4.30E-04	3.44E-06	7.91E-03	1.03E-04	9.02E+00
Vinyl chloride	8.34E-02	4.34E-03	1.71E-01	1.37E-03	6.68E-05	8.69E-07	5.45E+00

* Maximum ground level concentration (annual average) utilized (17-20 year period of operation)

RR = respiration rate for field vole (adult male) = 0.052 m3/d

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for field vole (40 g adult) = 0.008 kg/d

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

%G = percentage of vole diet as grazed vegetation (grass and leafy plants) - assumed to be 100%.

Flg = grazed vegetation ingestion rate for field vole (40 g adult) = 13 g/d (0.013 kg/d)

SI = soil ingestion rate for field vole = 0.13 g/d (1.3E-4 kg/d) - assumed to be 1% of Flg

***AF=1 - absorption assumed to be 100% where no data was available for chemical.

Life = average life expectancy for field vole = 90 days (3 mo.)

BW = average body weight (adult) = 40 g (0.04 kg)

Landfill-field vole model (2)

CHEMICAL	Soil Ingestion Dose SID = (Cs*SI) (ug/d)	Total Exposure Dose TED = (ID+WID+VID+SID) (ug/d)	Absorp. Factor AF (data input)***	Transfer Coefficient Fi = (AF*Life/BW) (d/kg)	Final Vole Concentration Cfvo = TED*Fi (ug/kg)	Final Vole Concentration (ppm) conversion = Cfvo/1000 (ug/g)
Benzene	3.55E-03	1.75E-02	1.00E+00	2.25E+03	3.93E+01	3.93E-02
Bromodichloromethane	7.50E-03	1.71E-02	1.00E+00	2.25E+03	3.85E+01	3.85E-02
1,2-dibromoethane	9.24E-03	9.61E-03	1.00E+00	2.25E+03	2.16E+01	2.16E-02
1,1-dichloroethane	2.12E-02	8.59E-02	1.00E+00	2.25E+03	1.93E+02	1.93E-01
1,2-dichloroethane	2.45E-04	7.74E-04	1.00E+00	2.25E+03	1.74E+00	1.74E-03
1,1-dichloroethylene	5.73E-03	8.09E-03	1.00E+00	2.25E+03	1.82E+01	1.82E-02
Ethyl mercaptan	0.00E+00	9.15E-03	1.00E+00	2.25E+03	2.06E+01	2.06E-02
Methyl mercaptan	0.00E+00	1.41E-02	1.00E+00	2.25E+03	3.18E+01	3.18E-02
Methylene chloride	4.97E-02	1.12E-01	1.00E+00	2.25E+03	2.51E+02	2.51E-01
1,1,2,2-tetrachloroethane	1.53E-04	1.47E-03	1.00E+00	2.25E+03	3.30E+00	3.30E-03
1,12-trichloroethylene	2.01E-03	2.63E-03	1.00E+00	2.25E+03	5.92E+00	5.92E-03
Trichloroethylene	1.17E-03	8.45E-03	1.00E+00	2.25E+03	1.90E+01	1.90E-02
Vinyl chloride	7.08E-04	6.41E-03	1.00E+00	2.25E+03	1.44E+01	1.44E-02

Landfill-vole-fox model (2)

CHEMICAL	Ground Level concentration GLC (data input)* (ug/m3)	Inhalation Dose ID = (GLC*RR) (ug/d)	Water Concentration Cw (data input)** (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW) (ug/kg)	Final Vole Concentration Cfvo (data input) (ug/kg)	Vole Ingest. Dose VolD = (Cfv*FI) (ug/d)	Soil Concentration Cs (data input) (ug/kg)
Benzene	2.62E-01	5.23E-01	2.60E-02	1.09E-02	3.93E+01	1.37E+01	2.73E+01
Bromodichloroethane	1.78E-01	3.56E-01	3.10E-02	1.30E-02	3.85E+01	1.35E+01	5.77E+01
1,2-dibromoethane	6.48E-03	1.30E-02	0.00E+00	0.00E+00	2.16E+01	7.57E+00	7.11E+01
1,1-dichloroethane	5.04E-01	1.01E+00	4.80E+00	2.02E+00	1.93E+02	6.77E+01	1.63E+02
1,2-dichloroethane	5.82E-03	1.16E-02	2.80E-02	1.18E-02	1.74E+00	6.10E-01	1.89E+00
1,1-dichloroethylene	3.88E-02	7.76E-02	4.20E-02	1.76E-02	1.82E+01	6.37E+00	4.41E+01
Ethyl mercaptan	1.76E-01	3.52E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methyl mercaptan	2.72E-01	5.44E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.18E+00	2.36E+00	5.00E-02	2.10E-02	2.51E+02	8.79E+01	3.82E+02
1,1,2,2-tetrachloroethane	1.80E-02	3.60E-02	1.80E-02	7.56E-03	3.30E+00	1.16E+00	1.18E+00
1,1,2-trichloroethane	6.98E-03	1.40E-02	3.00E-02	1.26E-02	5.92E+00	2.07E+00	1.55E+01
Trichloroethylene	1.38E-01	2.76E-01	4.30E-04	1.81E-04	1.90E+01	6.66E+00	9.02E+00
Vinyl chloride	8.34E-02	1.67E-01	1.71E-01	7.18E-02	1.44E+01	5.05E+00	5.45E+00

* Maximum ground level concentration (annual average) utilized (17-20 year period of operation).

RR = respiration rate for red fox (5 kg adult) = 2 m3/d

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for red fox (5 kg adult) = 420 g/d (0.42 kg/d)

%SW = percent of water ingested from impacted water body -assumed to be 100% in this case.

FI = vole ingestion rate for red fox (non-breeding 5 kg adult) = 350 g/d (0.35 kg/d)

SI = soil ingestion rate for red fox = 3.53 g/d (0.0035 kg/d) - assumed to be 1% of FI.

*** AF=1 - absorption assumed to be 100% where no data was available for a chemical.

Life = average life expectancy for red fox = 548 days (1.5 yrs).

BW = average body weight for red fox (adult male) = 5.0 kg

Landfill-vole-fox model (2)

CHEMICAL	Soil Ingestion Dose SID = (Cs*SI) (ug/d)	Total Exposure Dose TED = (ID+WID+VoID+SID) (ug/d)	Absorp. Factor AF (data input)***	Transfer Coefficient Fi = (AF*Life/BW) (kg/d)	Final Fox Concentration Cff = TED*Fi (ug/kg)	Final Fox Concentration (ppm) conversion = Cff/1000 (ug/g)
Benzene	9.56E-02	1.44E+01	1.00E+00	1.10E+02	1.58E+03	1.58E+00
Bromodichloroethane	2.02E-01	1.41E+01	1.00E+00	1.10E+02	1.54E+03	1.54E+00
1,2-dibromoethane	2.49E-01	7.83E+00	1.00E+00	1.10E+02	8.58E+02	8.58E-01
1,1-dichloroethane	5.72E-01	7.13E+01	1.00E+00	1.10E+02	7.81E+03	7.81E+00
1,2-dichloroethane	6.60E-03	6.40E-01	1.00E+00	1.10E+02	7.01E+01	7.01E-02
1,1-dichloroethylene	1.54E-01	6.62E+00	1.00E+00	1.10E+02	7.26E+02	7.26E-01
Ethyl mercaptan	0.00E+00	3.52E-01	1.00E+00	1.10E+02	3.86E+01	3.86E-02
Methyl mercaptan	0.00E+00	5.44E-01	1.00E+00	1.10E+02	5.96E+01	5.96E-02
Methylene chloride	1.34E+00	9.16E+01	1.00E+00	1.10E+02	1.00E+04	1.00E+01
1,1,2,2-tetrachloroethane	4.12E-03	1.20E+00	1.00E+00	1.10E+02	1.32E+02	1.32E-01
1,1,2-trichloroethane	5.42E-02	2.15E+00	1.00E+00	1.10E+02	2.36E+02	2.36E-01
Trichloroethylene	3.16E-02	6.97E+00	1.00E+00	1.10E+02	7.63E+02	7.63E-01
Vinyl chloride	1.91E-02	5.31E+00	1.00E+00	1.10E+02	5.82E+02	5.82E-01

Landfill-vole-hawk model

CHEMICAL	Ground Level concentration GLC (data input)* (ug/m3)	Inhalation Dose ID = (GLC*RR) ug/d)	Water Concentration Cw (data input)** (ug/kg)	Water Ingest. Dose WID = (Cw*WI*%SW) (ug/d)	Final Vole Concentration Cfvo (data input) (ug/d)	Vole Ingest. Dose VoID = (Cfv*FI) (ug/d)	Soil Concentration Cs (data input) (ug/kg)
Benzene	2.62E-01	4.71E-01	2.60E-02	1.82E-03	3.93E+01	4.71E+00	2.73E+01
Bromodichloroethane	1.78E-01	3.20E-01	3.10E-02	2.17E-03	3.85E+01	4.62E+00	5.77E+01
1,2-dibromoethane	6.48E-03	1.17E-02	0.00E+00	0.00E+00	2.16E+01	2.59E+00	7.11E+01
1,1-dichloroethane	5.04E-01	9.07E-01	4.80E+00	3.36E-01	1.93E+02	2.32E+01	1.63E+02
1,2-dichloroethane	5.82E-03	1.05E-02	2.80E-02	1.96E-03	1.74E+00	2.09E-01	1.89E+00
1,1-dichloroethylene	3.88E-02	6.98E-02	4.20E-02	2.94E-03	1.82E+01	2.18E+00	4.41E+01
Ethyl mercaptan	1.76E-01	3.17E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methyl mercaptan	2.72E-01	4.90E-01	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Methylene chloride	1.18E+00	2.12E+00	5.00E-02	3.50E-03	2.51E+02	3.01E+01	3.82E+02
1,1,2,2-tetrachloroethane	1.80E-02	3.24E-02	1.80E-02	1.26E-03	3.30E+00	3.96E-01	1.18E+00
1,1,2-trichloroethane	6.98E-03	1.26E-02	3.00E-02	2.10E-03	5.92E+00	7.10E-01	1.55E+01
Trichloroethylene	1.38E-01	2.48E-01	4.30E-04	3.01E-05	1.90E+01	2.28E+00	9.02E+00
Vinyl chloride	8.34E-02	1.50E-01	1.71E-01	1.20E-02	1.44E+01	1.73E+00	5.45E+00

* Maximum ground level concentration (annual average) utilized (17-20 year period of operation).

RR = respiration rate for redtail hawk (adult female) = 1.8 m3/d

** Utilized predicted worst-case groundwater concentrations

WI = water ingestion rate for redtail hawk (1.2 kg adult) = 70 g/d (0.07 kg/d)

%SW = percentage of water ingested from impacted water body - assumed to be 100% in this case.

FI = vole ingestion rate for redtail hawk (1.2 kg adult) = 120 g/d (0.12 kg/d)

SI = soil ingestion rate for redtail hawk = 0.12 g/d (0.0012 kg/d) - assumed to be 1% of FI.

*** AF=1 - absorption assumed to be 100% where no data were available for a chemical.

Life = average life expectancy of redtail hawk = 3285 days (9 yrs).

BW = average body weight of redtail hawk (adult female) = 1.2 kg

Landfill-vole-hawk model

CHEMICAL	Soil Ingestion Dose SID = (Cs*SI) (ug/d)	Total Exposure Dose TED = (ID+WID+VoID+SID) (ug/d)	Absorp. Factor AF (data input)***	Transfer Coefficient Fi = (AF*Life/BW) (kg/d)	Final Hawk Concentration Cfh = TED*FI (ug/kg)	Final Hawk Concentration (ppm) conversion - Cfh/1000 (ug/g)
Benzene	3.28E-02	5.22E+00	1.00E+00	2.74E+03	1.43E+04	1.43E+01
Bromodichloroethane	6.92E-02	5.02E+00	1.00E+00	2.74E+03	1.37E+04	1.37E+01
1,2-dibromoethane	8.53E-02	2.69E+00	1.00E+00	2.74E+03	7.37E+03	7.37E+00
1,1-dichloroethane	1.96E-01	2.46E+01	1.00E+00	2.74E+03	6.75E+04	6.75E+01
1,2-dichloroethane	2.26E-03	2.24E-01	1.00E+00	2.74E+03	6.13E+02	6.13E-01
1,1-dichloroethylene	5.29E-02	2.31E+00	1.00E+00	2.74E+03	6.32E+03	6.32E+00
Ethyl mercaptan	0.00E+00	3.17E-01	1.00E+00	2.74E+03	8.67E+02	8.67E-01
Methyl mercaptan	0.00E+00	4.90E-01	1.00E+00	2.74E+03	1.34E+03	1.34E+00
Methylene chloride	4.59E-01	3.27E+01	1.00E+00	2.74E+03	8.96E+04	8.96E+01
1,1,2,2-tetrachloroethane	1.41E-03	4.32E-01	1.00E+00	2.74E+03	1.18E+03	1.18E+00
1,1,2-trichloroethane	1.86E-02	7.43E-01	1.00E+00	2.74E+03	2.03E+03	2.03E+00
Trichloroethylene	1.08E-02	2.54E+00	1.00E+00	2.74E+03	6.96E+03	6.96E+00
Vinyl chloride	6.54E-03	1.90E+00	1.00E+00	2.74E+03	5.20E+03	5.20E+00

APPENDIX D-III

SUMMARY TABLES OF TERRESTRIAL EFFECTS OF MERCURY

From: CCME Draft Report, 1994: Ecological Effects: Mercury

Appendix 3. Summary Tables of Terrestrial Effects of Mercury (extracted from CCME, 1994).

Table 8: Summary of the toxicological data on the effects of mercury to terrestrial invertebrates.

Hg Compound	Invertebrate Species	Exposure Period	Exposure Concentration	Test substrate	Endpoint	Effect concentration mg.kg^{-1}dw	Reference
$HgCl_2$	Earthworm (*Eisenia foetida)*	14 days	0.07 to 565 mg.kg^{-1} soil	Artifical soil: pH 4 to 4.2; sand 70 to 75%; clay 16 to 22%; silt 8 to 13%; moisture 80% WHC; org. matter 4.7 to 10.4%	NOEC (survival) LOEC (58% mortality) LC_{25} LC_{50}	96 194 130 181	Environment Canada 1995
$HgCl_2$	Earthworm *(Octochaetus pattoni)*	5-60 days	0, 0.5, 1.0,. 2.0, and 5.0 mg.kg^{-1} soil	soil mixed with animal dung	LC_{50} - 5d	> 5	Abbasi and Soni 1983
					LC_{50} - 10d	2.39	
					LC_{50} - 15d	1.66	
					LC_{50} - 30d	1.51	
					LC_{50} - 40d	1.04	
					LC_{50} - 50d	0.91	
					LC_{50} - 60d	0.79	
CH_3HgCl	Red wiggler *(Eisenia foetida savigny)*	12 weeks	0, 1, 5, 25, 125 mg.kg^{-1} soil w.w.	30 g potting soil in petri dishes	NOEC LOEC (21% mortality)	1 5	Beyer *et al.* 1985
$HgCl_2$	Slug *(Arion aeter)*	27 days	0, 10, 25, 50, 100, 300 and 1000 mg.kg^{-1} feed	0-1000 mg.kg^{-1} in diet	NOEC (food consumption)	10 mg.kg^{-1} feed no dose related effects up to 100 mg.kg^{-1}	Marigomez *et al.* 1986
CH_3HgOH (MMH)	Aphids *(Macrosiphum gei* Ashmead)	16 days	0, 0.004, 0.02, and 0.04 mg.L^{-1}	aphids fed on tomato seedlings grown in a culture solution with MMH	LOEC (repro-duction) LD_{50}	0.004 mg.L$_{-1}$ in treatment solution 25 mg.L$_{-1}$ (body burden)	Haney and Lipsey 1973

Table 9: Summary of the toxicological data on the effects of mercury to terrestrial plants.

Hg Compound	Exposure Period	Exposure Concentration	Plant Species	Test substrate	Endpoint	Effect concentration mg.kg^{-1}dw	Reference
$HgCl_2$	72 hours	0.07 to 605 mg.kg.^{1}	Radish *(Raphanus sativa)*	Artificial soil: pH 4 to 4.2; sand 70 to 75%; clay 16 to 22%; silt 8 to 13%; moisture 80% WHC; org. matter 4.7 to 10.4%	Seedling emergence		Environment Canada 1995
					NOEC	51	
					LOEC	103	
					EC_{25}	73	
					EC_{50}	103	
$HgCl_2$	72 hours	0.001 to 270 mg.L	Radish *Raphanus sativa)*	Nutrient solution	root elongation		
					NOEC	12 mg.L^{-1}	
					LOEC (25% reduction)	16 mg.L^{-1}	
					EC_{25}	15 mg.L^{1}	
					EC_{50}	70 mg.L^{1}	
$HgCl_2$	120 hours	0.07 to 51 mg.kg^{1}	Lettuce *(Lactuca sativa)*	Artificial soil: pH 4 to 4.2; sand 69 to 75%; clay 16 to 22%; silt 8 to 10%; moisture 80% WHC; org. matter 4.7 to 6.3%	seedling emergence		
					NOEC	7	
					LOEC (27% reduction)	12	
					EC_{25}	11	
					EC_{50}	15	
$HgCl_2$	120 hours	0.001 to 58 mg.L^{-1}	Lettuce *(Lactuca sativa)*	Nutrient solution	root elongation		
					NOEC	6 mg.L^{-1}	
					LOEC (50% reduction)	12 mgl.L^{1}	
					EC_{25}	9 mg.L^{-1}	
					EC_{50}	18 mg.L^{-1}	
$HgCl_2$	10 days	0, 1, 5, 10 mg Hg.L^{-1}	Pea *(Pisum sativum)* *Mentha spicata*	nutrient solution	NOEC	1 mg.L^{-1}	Beauford et al. 1977
					LOEC (shoot length, root length, plant growth)	5 mg.L^{-1}	
$HgCl_2$	24 hours	0, 1, 10, 100 mgHg.L^{-1}	Alfalfa *Medicago sativa)* Okra *Abelmoschus esculentus)*	nutrient solution	NOEC (chlorophyll)	1 mg.L^{-1}	Mhatre and Chaphekar 1984
					LOEC (chlorophyll)	10 mg.L^{-1}	
$HgCl_2$	24 hours	0, 1, 10, 100 mgHg.L^{-1}	Cereal *(Pennisetum typhoideum)*	nutrient solution	NOEC (leaf injury)	1 mg.L^{-1}	Mhatre and Chaphekar 1984
					LOEC (leaf injury)	10 mg.L^{-1}	
					NOEC (chlorophyll)	10 mg.L^{-1}	
					LOEC (chlorophyll)	100 mg.L^{-1}	
$HgCl_2$	6 weeks outdoor growth following a single application	0, 1, 8, 25, 50 mg kg^{-1} d.w.	Bermudagrass *(Cynodon dactylon)*	Black clay: O.M. 3.9%, clay 65%, pH 7.6	NOEC (plant growth)	50 mg.kg^{1}	Weaver et al. 1984
				Silt loam: O.M. 1.29%, clay 25%, pH 7.7	LOEC (plant growth)	8 mg.kg^{-1}	

Table 9: Summary of the toxicological data on the effects of mercury to terrestrial plants.

Hg Compound	Exposure Period	Exposure Concentration	Plant Species	Test substrate	Endpoint	Effect concentration mg.kg^{-1}dw	Reference
				Fine sand: O.M. 0.39%, clay 6%, pH 4.7	LOEC (plant growth)	50 mg.kg^{-1}	
$HgCl_2$	7 weeks	0, 0.2, 2.0, 200 mg.L^{-1}	Pine *(Picea abies)*	Nutrient solution	NOEC	2 mg.L^{-1}	Godbold 1991
					LOEC (root growth)	200 mg.L^{-1}	
CH_3HgOH	2-10 days	0, 0.05, 0.5, 1.0, 5, 25 mg Hg.L^{-1}	Tomato seedlings *(Lycopersicom esculentum)*	modified Hoagland's solution	LOEC (plant growth, tomato)	0.01 mg.kg^{-1}	Haney and Lipsey 1973
			Beans *(Phaseolus vulgaris)*	modified Hoagland's solution	LOEC (plant growth, beans)	0.06 mg.kg^{-1}	
PMA		455 mg.kg^{-1}	bent grass *Agrostis canina*	garden soil	no effects	no toxicity	Estes et al. 1973
$HgCl_2$	50-120 days	0, 10, 20, 46, 100, 220, 460 and 1000 mg.kg^{-1}	Turnip *(Brassica rapa)* Lettuce *(Lactuca sativa)*	clay soil (43% clay, pH 7.9, O.M. 27 g.kg^{-1}) sand (3% clay, pH 6.3, O.M. negligible) and garden soil: commercial potting soil	reduced bloom initiation	10-22 mg for sand, 220 mg for clay and pot soil	Sheppard et al. 1993
Hg^0 vapour	7 and 14 days	NA	15 test species	Air saturation of 14 ug.L^{-1}	NOEC (seed germination and early seedling growth)	14 ug.L^{-1}	Siegel et al. 1984

Table 10 Summary of the acute toxicological data of the effects of mercury on terrestrial birds

Hg Compound	Species	Exposure Period	Exposure Route	Effect[1]	Effect Concentration (mg.kg^{-1}bw)	Reference
CH_3HgCl	Japanese Quail	acute[2]	oral	LD_{50}	18	Hill and Soares 1984
$HgCl_2$		acute	oral	LD_{50}	42	
CH_3HgCl		5 days	oral	LC_{50}	47	
$HgCl_2$		5 days	oral	LC_{50}	5086	
MeHg[3]		5 days	oral	LC_{50}	~ 1750	Hill and Shaffner 1975
MeHg[4]		acute	oral	LD_{50}	668	Hudson et al. 1984
MeHg[5]		acute	oral	LD_{50}	1498	
PMA[7]		5 days	oral	LC_{50}	614	Hill and Camardese 1986
MeHg[6]		5 days	oral	LC_{50}	45	
MeHg[4]		5 days	oral	LC_{50}	147	
MeHg[4]	Pheasant	acute	oral	LD_{50}	360	Hudson et al. 1984
MeHg[5]		acute	oral	LD_{50}	1190	
PMA[7]		acute	oral	LD_{50}	169	
$HgCl_2$		5 days	oral	LC_{50}	3790	Hill and Shaffner 1975
MeHg[3]		5 days	oral	LC_{50}	1102	
PMA		5 days	oral	LC_{50}	~ 2350	Hill and Shaffner 1975
MeHg[4]		acute	oral	LC_{50}	64	
MeHg[4]		acute	oral	LC_{50}	146	
MeHg[4]	Mallard Duck	acute	oral	LD_{50}	2262	Hudson et al. 1984
MeHg[5]	Mallard Duck	acute	oral	LD_{50}	2000	Hudson et al. 1984
PMA[7]		acute	oral	LD_{50}	878	
$HgCl_2$	Mallard Duck	5 days	oral	LC_{50}	> 5000	Hill and Shaffner 1975
MeHg[3]		5 days	oral	LC_{50}	~ 280	
PMA		5 days	oral	LC_{50}	~ 1175	

Table 10 Summary of the acute toxicological data of the effects of mercury on terrestrial birds

Hg Compound	Species	Exposure Period	Exposure Route	Effect[1]	Effect Concentration (mg.kg^{-1}bw)	Reference
MeHg[6]	Bobwhite Quail	5 days	oral	LC_{50}	60	
MeHg[4]		5 days	oral	LC_{50}	~ 50	
MeHg[5]		acute	oral	LD_{50}	1060	Hudson et al. 1984
MeHg[4]		5 days	oral	LC_{50}	~ 70	Hill and Shaffner 1975
MeHg[4]	Prairie Chicken	acute	oral	LD_{50}	360	Hudson et al. 1984
MeHg[4]	Chukar Partridge	acute	oral	LD_{50}	841	
MeHg[4]	Grey Partridge	acute	oral	LD_{50}	550	
MeHg[4]	Rock Dove	acute	oral	LD_{50}	714	
MeHg[4]	Fulvous Whistling Duck	acute	oral	LD_{50}	1680	Hudson et al. 1984

1. LC_{50} expressed as mg compound.kg^{-1} body wt.
2. acute as single oral dose
3. methoxyethylmercury
4. Ceresan M
5. Ceresan L
6. Morsodren
7. Phenylmercuric acetate

Table 11 Summary of the subacute toxicological data on the effects of mercury on terrestrial birds

Hg Compound	Species	Exposure Period	Exposure Concentration	Exposure Route	Endpoint/Effect	Effect Concentration	Reference
Ch_3Hg^1	Mallard	21 weeks	0.5 and 3 $mg.kg^{-1}$ d.w. in feed	oral	decreased egg laying and # hatching, increased mortality of hatchlings	3 $mg.kg^{-1}$	Heinz 1974
MMD[2]	White Leghorn Chicks	21 days	6,12,18 $mg.kg^{-1}$	oral	LC_8	18 $mg.kg^{-1}$	Fimreite 1970
MMD	Red-Tailed Hawks	4-12 weeks	contaminated chicks: 3.9, 7.2 and 10 $mg.kg^{-1}$	oral	neurotoxicity death	10 $mg.kg^{-1}$	Fimreite and Karstad 1971
Ch_3Hg^1	Mallard Ducks	2 genera- tions	0.5 and 3 $mg.kg^{-1}$	oral	decreased growth and survival of hatchlings	3 $mg.kg^{-1}$	Heinz 1976
$HgCl_2$	Japanese Quail	1 year	2,4,8,16 $mg.kg^{-1}$	oral	NOEC	32 $mg.kg^{-1}$	Hill and Shaffner 1976
PMA[3]	Pheasants	--	20 mg Hg kg^{-1}	oral	decreased hatchling, growth and survival of chicks	20 mg $Hg.kg^{-1}$	Mullens et al 1977
CH_2Hg	Chicken eggs	--	--	yolk injection	TL_{50}[4]	0.1-0.5 $mg.kg^{-1}$	Birge and Roberts 1976
$HgCl_2$					TL_{50}	1.0 mg kg^{-1}	
CH_3HgCl	Chickens	3 weeks	10,20 $mg.kg^{-1}$	oral	decreased growth and decreased hatching numbers	10 $mg.kg^{-1}$	Scott 1977
CH_3Hg	Black Ducks	28 weeks	3 $mg.kg^{-1}$ (single dose)	oral	hatchling and duckling mortality (LD_{75})	3 $mg.kg^{-1}$	Finley and Stendell 1988
CH_3HgCl	Mallard eggs	10 days	50, 500 $mg.kg^{-1}$	surface applica-tion to egg	decreased embryonic growth	50 $mg.kg^{-1}$	Hoffman and Eastin 1981
CH_3HgCl	Pekin Ducks	12 weeks	0.5,5,15.0 $mg.kg^{-1}$	oral	NOEC growth and liver damage LOEC	0.48 $mg.kg^{-1}$ 3.76 $mg.kg^{-1}$	Bhatnagar et al. 1982
CH_3HgCl	Chicken eggs			yolk injec-tion	LD_{50}	40-50 $mg.kg^{-1}$	Greener and Kochen 1983
CH_3HgCl	Bobwhite Quail	6 weeks	5.4,20.0 $mg.kg^{-1}$	oral	NOEC mortality	5.4,20.0 $mg.kg^{-1}$	Spann et al. 1986
CH_3HgCl	Zebra Finches	76 days	1.0,2.5.5.0 $mg.kg^{-1}$	oral	NOEC Neurotoxicity and mortality	2.5,5.0 $mg.kg^{-1}$	Scheuhammer 1988

1. Morsodren (commercial product)
2. MMD = methyl mercuric dicyandiamide
3. PMA = Phenyl mercuric acetate
4. Tl_{50} - 50 percent survival toxicant level

Table 12 Summary of available toxicological data on the effects of mercury on mammals

Hg Compound	Species	Exposure Period	Exposure Concentration	Exposure Route	Effect	Effect Concentration	Reference
MeHgCl	Rats (31 to 41 g weight)	2 and 12 weeks	0.0, 0.0, 0.4, 2, 10, 20 and 200 mg.kg^{-1} bw	oral	NOEC LOEC (toxic signs, weight loss and food intake)	10 mg.kg^{-1} 20 mg.kg^{-1}	Verschuuren et al. 1976a
MeHgCl	Rats (46 to 50 g weight)	2 years	0.0, 0.08, 0.4, and 2 mg.kg^{-1} bw	oral	NOEL LOEC* (growth reduction, kidney damage and histochemical changes)	0.08 to 0.4 mg.kg^{-1} bw 2 mg.kg^{-1} bw	Verschuuren et al. 1976b
CH_3HgCl	Cat	2 years	0,3,8.4,20,46,74,176 ug Hg.(kg.day)$^{-1}$	oral	NOEL LOEL	20 ug kg^{-1} b.w. 46 ug.kg^{-1} b.w.	Charbonneau et al. 1976
CH_3HgCl	Mouse	2 years	0,0.4,2,10 mg.kg^{-1} feed	oral	NOEL LOEC (chronic nephropathy)	0.4 mg.kg^{-1} feed 2 mg.kg^{-1} feed	Mitsumori et al. 1990
MMD	Pigs	single exposure	0.0, 1.7, 3.4,6.7,10.1, 13.4, 53.8, 26.9 and 107.4 mg.kg^{-1} bw	oral	NOEL LOED	1.7 mg.kg^{-1} feed 13.4 mg.kg^{-1} feed	Piper et al. 1971
MMD and EMC	Pigs	60 to 90 days	0.0, 0.19, 0.38 and 0.76 mg.kg^{-1} bw	oral	NOED LOED	0.19 mg.kg^{-1} 0.38 mg.kg^{-1}	Tryphonas and Nielsen 1973
$HgCl_2$	Mice	single exposure	0, 25, 50 and 200 umol.kg^{-1}	oral	NOED LOED (lipid peroxidase)	50 umol.kg^{-1} 200 umol.kg^{-1}	Nielsen et al. 1991
CH_3Hg_0	Rhesus Monkey	1 year	80 to 125 ug kg^{-1} bw	oral	NOEL	125 ug.kg^{-1} b.w.	Chen et al. 1983

MMD = Methylmercuric dycyandiamide; EMC = Ethylmercuric chloride

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APPENDIX D-IV

SUMMARY TABLES OF TERRESTRIAL EFFECTS OF BENZENE

From: CCME Draft Report, 1994: Ecological Effects: Benzene

TABLE 4: Summary of toxicological date for benzene (extracted from CCME, 1994).

Species	Exposure Period	Chemical Used	Test Substrate	Endpoint	Effect Concentration (mg.kg^{-1})	Reference
AGRICULTURAL CROPS						
Radish (*Raphanus sativa)*	3 days	benzene	50% artificial soil, 50% sand; 80% moisture with deionized water	NOEC (seedling emergence) LOEC (70% reduction in seedling emergence EC_{25} - by linear interpolation (reduction in seedling emergence) EC_{50} - by linear interpolation	548 (applied) 16 (recovered) 1096 (applied) 32 (recovered) 783 (applied) 22 (recovered) 961 (applied) 27 (recovered)	Environment Canada, 1995
Radish (*Raphanus sativa)*	3 days	benzene	artificial soil topped with filter; filter hydrated with 2 mL DRO; 100% moisture	NOEC (root length) LOEC (61% reduction in root length) EC_{25} - by linear interpolation (reduction in rooth length) EC_{50} - by linear interpolation (reduction in root length)	1096 (applied) 117 (recovered) 2191 (applied) 234 (recovered) 1187 (applied) 127 (recovered) 1886 (applied) 201 (recovered)	Environment Canada, 1995
Radish (*Raphanus sativa)*	3 days	benzene	artificial soil topped with filter; filter hydrated with 2 mL DRO; 100% moisture	NOEC (root length) LOEC (23% reduction in root length) EC_{25} - by linear interpolation (reduction in root length) EC_{50} - by linear interpolation (reduction in root length)	274 (applied) 8 (recovered) 548 (applied) 16 (recovered) 467 (applied) 14 (recovered) 1333 (applied) 39 (recovered)	Environment Canada, 1995

TABLE 4: Summary of toxicological date for benzene.

Species	Exposure Period	Chemical Used	Test Substrate	Endpoint	Effect Concentration (mg.kg^{-1})	Reference
Radish *(Raphanus sativa)*	3 days	benzene	60% artificial soil; 40% sand; 80% moisture	NOEC (seedling emergence)	137 (applied) 15 (recovered)	Environment Canada, 1995
				LOEC (33% reduction in seedling emergence)	274 (applied) 29 (recovered)	
				EC$_{25}$ - by linear interpolation (reduction in seedling emergence)	240 (applied) 26 (recovered)	
				EC$_{50}$ - by linear interpolation (reduction in seedling emergence)	959 (applied) 102 (recovered)	
Lettuce *(Lactuca sativa)*	5 days	benzene	artificial soil topped with filter; filter hydrated with 2 mL DRO; 100% moisture	NOEC (root length)	1096 (applied) 32 (recovered)	Environment Canada, 1995
				LOEC (77% reduction in root length)	2191 (applied) 64 (recovered)	
				EC$_{25}$ - by linear interpolation (reduction in root length)	1031 (applied) 30 (recovered)	
				EC$_{50}$ - by linear interpolation (reduction in root length)	1562 (applied) 46 (recovered)	
Lettuce *(Lactuca sativa)*	5 days	benzene	50% artificial soil; 50% sand; 80% moisture	NOEC (seedling emergence)	137 (applied) 4 (recovered)	Environment Canada, 1995
				LOEC (33% reduction in seedling emergence)	274 (applied) 8 (recovered)	
				EC$_{25}$ - by linear interpolation (reduction in seedling emergence)	240 (applied) 7 (recovered)	
				EC$_{50}$ - by linear interpolation (reduction in seedling emergence)	890 (applied) 26 (recovered)	

TABLE 4: Summary of toxicological date for benzene.

Species	**Exposure Period**	**Chemical Used**	**Test Substrate**	**Endpoint**	**Effect Concentration ($mg.kg^{-1}$)**	**Reference**
Lettuce (*Lactuca sativa)*	5 days	benzene	60% artificial soil; 40% sand, 80% moisture	NOEC (seedling emergence)	548 (applied) 58 (recovered)	Environment Canada, 1995
				LOEC (67% reduction in seedling emergence)	1096 (applied) 117 (recovered)	
				EC_{25} - by linear interpolation (reduction in seedling emergence)	685 (applied) 73 (recovered)	
				EC_{50} - by linear interpolation (reduction in seedling emergence)	932 (applied) 99 (recovered)	
Lettuce (*Lactuca sativa)*	5 days	benzene	artificial soil topped with filter; filter hydrated with 2 mL DRO; 100% moisture	NOEC (root length)	274 (applied) 29 (recovered)	Environment Canada, 1995
				LOEC (54% reduction in root length)	1096 (applied) 117 (recovered)	
				EC_{25} - by linear interpolation (reduction in root length)	527 (applied) 56 (recovered)	
				EC_{50} - by linear interpolation (reduction in root length)	970 (applied) 103 (recovered)	
INVERTEBRATES						
Earthworm (*Eisenia foetida)*	2 days	benzene	filter paper	LC_{50}	98 $ug.cm^{-2}$	Newhauser et al., 1986
Earthworm *Eisenia foetida)*	2 days	benzene	filter paper	LC_{50}	100-1000 $ug.cm^{-2}$	Roberts and Dorough, 1984
Earthworm (*Eisenia foetida)*	6 weeks	benzene	4 mm layer of silt loam in petri dish	NOEC (growth)	4% (w/w) dissolved in 30 g sludge with 13% solids	Hartenstein, 1982
				LOEC (growth)	8% (w/w) same as above	
Mosquito (*Aedes aegypt)*	24 hours	benzene	glass bowls of distilled water	LC_{50}	59.3 $mg.L^{-1}$	Berry and Brammer, 1977
				non-lethal dose	12.8 $mg.L^{-1}$	

TABLE 4: Summary of toxicological date for benzene.

Species	Exposure Period	Chemical Used	Test Substrate	Endpoint	Effect Concentration (mg.kg^{-1})	Reference
Nematode *(Panagrelius redivivus)*	4 days	benzene	methanol extract of artificial soil (70% industrial sand; 20% clay; 10% sphagnum peat moss)	NOEC (survival)	7.8 mg.L^{-1}	Samolloff et al., 1980
				NOEC (growth)	7.8 mg.L^{-1}	
				NOEC (maturation)	0.78 mg.L^{-1}	
Earthworm *(Eisenia foetida)*	14 days	benzene	artificial soil; 80% moisture	NOEC (mortality)	2191 (applied) 234 (recovered)	Environment Canada, 1995
				LOEC (100% mortality	4383 (applied) 467 (recovered)	
				LC$_{25}$ - by linear interpolation	2622 (applied) 280 (recovered)	
				LC$_{50}$ - by linear interpolation	3209 (applied) 342 (recovered)	
Earthworm *(Eisenia foetida)*	14 days	benzene	artificial soil; 80% moisture	NOEC (mortality)	1096 (applied 32 (recovered)	Environment Canada, 1995
				LOEC (60% mortality)	2191 (applied) 64 (recovered)	
				LC$_{25}$ - by linear interpolation	1440 (applied) 42 (recovered)	
				LC$_{50}$ - by linear interpolation	1880 (applied) 55 (recovered)	
MAMMALS						
Rat	4 hours	benzene	inhalation	LC$_{50}$	41.6 g.m^{-3}	Mallinckrodt Inc., 1989
	96 hours		ingestion	LD$_{50}$	3306 mg.kg^{-1} bw	

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TECHNICAL REPORT– PART "E"
GLOSSARIES

I. GLOSSARY - HUMAN HEALTH RISK ASSESSMENT

Acute exposure - One dose or multiple doses occurring within a short time (24 hours or less)

Bioassay - The determination of the potency (bioactivity) or concentration of a test substance by noting its effects in live animals or in isolated organ preparations, as compared with the effect of a standard preparation

Bioavailability - The degree to which a drug or other substance becomes available to the target tissue after administration or exposure

Carcinogen - An agent capable of inducing a cancer response

Carcinogenesis - The origin or production of cancer, very likely a series of steps. The carcinogenic event so modifies the genome and/or other molecular control mechanisms in the target cells that these can give rise to a population of altered cells

Chronic effect - An effect that is manifest after some time has elapsed from initial exposure

Chronic exposure - Multiple exposures occurring over an extended period of time, or a significant fraction of the animal's or the individual's life-time

Critical effect - The first adverse effect, or its known precursor, that occurs as the dose rate increases

Deterministic Assessment - Toxicological health risk assessment using single values to characterize the variables used in the calculations (*e.g.,* body weight is 70 kg). The final output is a single value of potential health risks (*e.g.,* 1×10^{-6})

Developmental toxicity - The study of adverse effects on the developing organism (including death, structural abnormality, altered growth, or functional deficiency) resulting from exposure prior to conception (in either parent), during prenatal development, or postnatally up to the time of sexual maturation

Dose-response relationship - A relationship between the amount of an agent (either administered, absorbed, or believed to be effective) and changes in certain aspects of the biological system (usually toxic effects), apparently in response to that agent

Endpoint - A response measure in a toxicity study

Extrapolation - An estimation of a numerical value of an empirical (measured) function at a point outside the range of data which were used to calibrate the function. The quantitative risk estimates for carcinogens are generally low-dose extrapolations based on observations made at higher doses. Generally one has a measured dose and measured effect

Individual risk - The probability that an individual person will experience an adverse effect. This is identical to population risk unless specific population subgroups can be identified that have different (higher or lower) risks. In the present case, LECR estimated for receptors assumed to live during 30 years at the location corresponding to the maximum GLC

Lifetime Excess Cancer Risk (LECR) - Probability of developing cancer following lifetime exposure to a carcinogenic agent emitted by the source being investigated. It is an additional or an excess risk (*i.e.,* contribution from background levels not factored)

Maximal Exposed Individuals - Human receptors assumed to live at the point of maximum impact of the emissions (or location associated with maximum ground-level concentrations)

Population Risk - LECR estimated for an entire population exposed to varying concentrations of chemicals and assumed to be living during 30 years in the 20 km x 20 km area surrounding the sources

Probabilistic Assessment - Toxicological health risk assessment using ranges of values or statistical distributions to characterize the variables used in the calculations (*e.g.,* body weight for the adult group is represented by a normal distribution with a mean of 70 kg, standard deviation of 12 kg, minimum of 34 kg and maximum of 110 kg). The final output is a distribution of values of potential health risks (*e.g.,* mean: 1×10^{-6}; 95th percentile: 1×10^{-5}). The probabilistic assessment is conducted by using statistical sampling such as Monte-Carlo or Latin-Hypercube

Reference Concentration (RfC) - An estimate (with uncertainty spanning perhaps an order of magnitude) of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious noncancer effects during a lifetime

Reference Dose (RfD) - An estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime

Risk - The probability of injury, disease, or death under specific circumstances. In quantitative terms, risk is expressed in values ranging from zero (representing the certainty that harm will not occur) to one (representing the certainty that harm will occur). The following are examples showing the manner in which risk is expressed in IRIS: E-4 = a risk of 1/10,000; E-5 = a risk of

1/100,000; E-6 = a risk of 1/1,000,000. Similarly, 1.3E-3 = a risk of 1.3/1000 = 1/770; 8E-3 = a risk of 1/125; and 1.2E-5 = a risk of 1/83,000

Risk assessment - The determination of the kind and degree of hazard posed by an agent, the extent to which a particular group of people has been or may be exposed to the agent, and the present or potential health risk that exists due to the agent

Systemic Effects - In this context was used to describe any adverse effect not associated with cancer (*e.g.,* effects on liver, lung, blood, etc..)

Threshold - The dose or exposure below which a significant adverse effect is not expected. Carcinogens are thought to be non-threshold chemicals, to which no exposure can be presumed to be without some risk of adverse effect

Threshold Limit Values (TLVs) - Recommended guidelines for occupational exposure to airborne contaminants published by the American Conference of Governmental Industrial Hygienists (ACGIH). The TLVs represent the average concentration (in mg/m^3) for an 8-hour workday and a 40-hour work week to which nearly all workers may be repeatedly exposed, day after day, without adverse effect

Uncertainty factor - One of several, generally 10-fold factors, used in operationally deriving the Reference Dose (RfD) from experimental data. UFs are intended to account for (1) the variation in sensitivity among the members of the human population; (2) the uncertainty in extrapolating animal data to the case of humans; (3) the uncertainty in extrapolating from data obtained in a study that is of less-than-lifetime exposure; and (4) the uncertainty in using LOAEL data rather than NOAEL data

Unit Risk - The upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent at a concentration of 1 ug/L in water, or 1 ug/m^3 in air

Upper bound - An estimate of the plausible upper limit to the true value of the quantity. This is usually not a statistical confidence limit

II. GLOSSARY - ECOLOGICAL RISK ASSESSMENT

Acute Toxicity - An adverse response, ranging from injury to death, following exposure of short duration relative to the normal lifespan of the organism.

Additive Effects - Toxic interactions of two or more substances on organisms producing a result such that the total effect approximately equals the sum of the individual effects.

Aesthetic - Dealing with those aspects of water that are perceived by the sense's (e.g. taste or odour)

Application Factor (AF) - A numerical, unitless value, calculated as the threshold concentration of a chemical for chronic effects divided by its threshold concentration for acute effects. An AF is generally calculated by dividing the limits (no-observed-effect concentration [NOEC] and low-observed-effect concentration [LOEC]) of the maximum acceptable toxicant concentration (MATC) by the LC_{50}. The AF is usually reported as a range and is multiplied by the median lethal concentration of a chemical as determined in a short-term (acute) toxicity test to estimate an expected no-effect concentration for chronic exposure.

Assessment endpoint - The characteristic of the ecological system that is the focus of the risk assessment.

Assimilative Capacity - the limit of a water body to transform and/or incorporate substances (e.g. nutrients) by the ecosystem, such that the water quality does not degrade below a predetermined level.

Background Water Quality - The physical, chemical and/or biological conditions of a water body at a point upstream/up current of a polluting source. The establishment of background water quality may be based on historical data or on water quality in a similar, unaltered water body.

Benchmark Concentration (BC) - Specific concentrations at which some level Of effects is expected (e.g., LC_{25}, MATC). These concentrations are derived from the hazard assessment.

Best Available Treatment Technology - can include, in order of preference - changes in production processes, chemical substitution, in-plant controls, best management practices, water conservation, waste treatment technologies, energy conservation.

Bioaccumulation Factor (BAF) - The ratio of the concentration of a compound in the tissues of aquatic organisms relative to the water they live in.

Bioaccumulation - the process by which chemicals are amassed by organisms from water directly or through the consumption of food containing the chemicals.

Biomarkers - Biochemical or cellular indicators of exposure (e.g., body burdens, indicators of DNA damage, enzyme activity, and biochemical indicators Of reproductive or bioenergetic status).

Contaminant - A substance which, once in the water, may pose a threat to the ecosystem and/or human health, as well as uses such as water supply, recreation, and aesthetic conditions.

Conventional Pollutants - Traditional indicators of environmental quality including BOD, nutrients and solids. Materials defined as hazardous substances are not included in this category.

Ecological Risk Assessment (ERA)

•The process that evaluates the likelihood that adverse ecological effects may occur or occurring as a result of exposure to one or more stressors. This definition recognizes that a risk does not exist unless (1) the stressor has an inherent ability to cause adverse effects and (2) it is coincident with or contacts the ecological component long enough and at sufficient intensity to elicit the identified adverse effect(s).

•A subcategory of ecological impact assessment that (1) predicts the probability of adverse effects occurring in an ecosystem or any part of an ecosystem as a result of perturbation and (2) relates the magnitude of the impact to the perturbation.

•The process of assigning magnitudes and probabilities to adverse effects of human activities (or natural catastrophes).

•A formal set of scientific methods for estimating the probabilities and magnitudes of undesired effects on plants, animals, and ecosystems resulting from events in the environment, including the release of pollutants, physical modification of the environment, and natural disasters.

Effluent Limit - A legally enforceable effluent requirement.

Effluent - the wastewater discharge to a receiving water body.

Expected Environmental Concentration (EEC) - The calculated concentration of a chemical in a particular medium for a particular site.

Exposure - The process by which a chemical is delivered to an organism resulting in a dose (the amount of a chemical either in the organism as a whole or in a target tissue). Exposure is a result of the concentration and form of a chemical in the environment, coupled with the presence of the organism.

Exposure assessment - The process of estimating the dose received by an organism, population, or ecosystem. It may be prospective, in which case estimates of the chemical concentrations and forms in various media or habitats are combined with estimates of the organism's behaviour to predict dose. It may also be retrospective, in which case dose is estimated from body burdens of the chemical or changes in the organism caused by the chemical (biomarkers).

Hazard assessment - The overall process of evaluating the type and magnitude of adverse effects caused by a stressor.

Hazardous Substances - chemicals that are persistent, bioaccumulative and extremely toxic. They include substances which, individually or in combination with other substances, can cause death, disease including cancer, behavioural abnormalities, genetic mutations, physiological malfunctions and/or physical deformities.

Lowest-Observed-Effect Concentration (LOEC) - The lowest amount or concentration of a stressor for which some biological effect is observed.

Maximum Acceptable Toxicant Concentration (MATC) - The maximum concentration at which a stressor can be present and not be toxic to the test organism. The MATC is normally calculated as the geometric mean of the lowest concentration for which an adverse effect was observed (LOEC) and the highest concentration that did not yield any adverse effects (NOEC).

Measurement endpoint - An effect on an ecological component that can be measured and described in some quantitative fashion.

Median Effective Concentration (EC_{50}) - The concentration of a stressor in water that is estimated to be effective in producing some biological response, other than mortality, in 50% of the organisms over a specific time interval (e.g., a 48-h daphnid EC_{50}).

Median Lethal Concentration (LC_{50}) - The concentration of a stressor in water that is estimated to be lethal to 50% of the test organisms over a specific time interval (eg., a 96-h fish LC_{50}).

Modifying factor - Any characteristic of an organism or the surrounding environment that affects toxicity.

Monte Carlo simulation - An iterative process involving the random sampling of stochastic model parameter values from specified frequency distributions, simulation of the system, and output of predicted values. The distribution of the output values can be used to determine the probability of occurrence of any particular value.

Natural Background Water Quality - Chemical, physical and biological quality of a water body, if unaffected by human activity.

No-Observed-Effect Concentration (NOEC) - The amount or concentration of a stressor that does not result in any adverse effect.

Non-indigenous - Not native or not belonging naturally to an area.

Non-point Source - A non-specific or diffuse source entering the aquatic environment. Commonly, any source that cannot be described as a point source. Usually, this type of source is not amenable to collection and, if necessary, treatment.

Objectives - Provincial Water Quality Objectives.

Parameter - A measurable or quantifiable characteristic or feature of water quality.

Persistence - Longevity in the environment usually expressed as half-life.

Persistent Pollutant - A substance with an environmental half-life in excess of 50 days.

Point source - A source of pollution that is discharged to the environment at a specific location. It is quantitatively and qualitatively definable.

Pollution Prevention - Any action which eliminates or reduces the creation of pollutants or waste at their source through measures such as substitution/reduction in use of a raw product, production redesign, process change, in-process recycling and/or improved maintenance and operating procedures.

QSAR (quantitative structure activity relationship) - A method of estimating unmeasured physical and toxicological properties for a chemical on the basis of chemical structure, functional groups, and similarity to known chemicals.

Receiving Water - A water body to which a discharge is directed.

Receptor - The entity (e.g., organism, population, community, ecosystem) that might be adversely affected by contact with or exposure to a substance of concern.

Remedial Actions - Actions undertaken to upgrade water quality and restore uses to a water body previously impaired. Initial actions are usually to eliminate or reduce the source(s) of the problem and can include physical intervention (e.g., dredging, artificial aeration, flow augmentation, etc.)

Risk - The chance of an undesired effect, such as injury, disease, or death, resulting from human actions or a natural catastrophe.

Risk characterization - The evaluation of the likelihood that adverse ecological effects may occur as result of exposure to a stressor, including an evaluation of the consequences of these effects.

Risk assessment - A set of formal scientific methods for estimating the probabilities and magnitudes of undesired effects resulting from the release of chemicals, other human actions, or natural catastrophes.

Route of exposure (exposure pathway) - The means by which organisms are exposed to contaminants. Routes/pathways would include uptake of contaminants from solution, ingestion of contaminated food/prey, inhalation of contaminated particles, etc. More generally, routes of exposure include exposure via water, soil, sediments, food and other media.

Site characterization - Evaluation of available data and information concerning the site (e.g., site use, geology, hydrology, available chemistry, and toxicity data).

Surrogate - A convenient and/or practical substitute

Sustainability - The utilization of a resource in a manner that the resource will be adequately protected for indefinite future use.

Toxic substance - A substance capable of producing an adverse response, ranging from injury to death, in a living organism.

Toxicity - An adverse response, ranging from injury to death, in a living organism.

Valued Ecosystem Component (VEC) - Each of the environmental attributes or components identified as a result of societal values and considerations.

Waste - Any solid liquid, gas, odour, heat, sound, vibration, radiation or combination of any of them resulting directly or indirectly from the activities of man which may: impair the quality of the natural environment for any use that can be made of it; cause injury or damage to property or to plant or animal life; cause harm or material discomfort to any person; adversely affect the health or impair the safety of any person; or render any property or plant or animal life unfit for use by man.

Waste Assimilation - (See Assimilative capacity)

Water Conservation - Preservation of the quantity of available water through judicious use, reuse and minimal wastage.

Watershed - The area of land drained by a river/stream and its tributaries.